Urban flooding and health risk analysis by use of quantitative microbial risk assessment

Limitations and improvements

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Publication date:
2015

Document Version
Publisher's PDF, also known as Version of record

Technical University of Denmark

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-Limitations and Improvements

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DTU Environment
Department of Environmental Engineering

PhD Thesis
March 2015

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The synopsis part of this thesis is available as a pdf-file for download from the DTU research database ORBIT: http://www.orbit.dtu.dk

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Printed by:	Vester Kopi March 2015
Cover:	Torben Dolin

Preface

This thesis is the closing work of my PhD project, which was carried out at the Department of Environmental Engineering (DTU Environment), Technical University of Denmark (DTU) and the Danish hydraulic Institute (DHI). The thesis is based on the scientific results achieved during the PhD project in the period from December 2009 to November 2014. The project was supervised by Professor Hans-Jørgen Albrechtsen and Head of Research and Development Dr. Ole Mark. The PhD project was funded by DTU and The Danish Council for Strategic Research as a part of the Storm- and Wastewater Informatics (SWI) project. The PhD project was included in the Urban Water Technology Graduate School (UrbanWaterTech, DTU).

The thesis is organized in two parts: the first part sets the findings of the PhD into context in an introductive review; the second part consists of the papers listed below. These will be referred to in the text by their paper number written with the Roman numerals **I-IV**. The included papers are:

I **Andersen, S. T.**, Mark O., Sharma, A. K., Schultz, A. C., Albrechtsen, H.-J. Diurnal variations of *E. coli*, *Enterococci spp*. and Norovirus in wastewater at different catchments at dry weather flow: measured and modelled. *Submitted*.

II **Andersen, S. T.**, Mark O., Albrechtsen, H.-J. Survival of microorganisms in urban floodwater from overloaded combined sewer systems – effects of UV irradiation simulating sunlight. *Submitted*.

III **Andersen, S. T.**, Erichsen, A. C., Mark O., Albrechtsen, H.-J. (2013) Effects of a 20 year rain event: a quantitative microbial risk assessment of a case of contaminated bathing water in Copenhagen, Denmark. *Journal of Water and Health*, Vol. 11 (4), p. 636-646.

IV **Andersen, S. T.**, Mark O., Schultz, A. C., Albrechtsen, H.-J. Improvements of quantitative microbial risk assessment of urban flooding by combining quantitative microbial data with a drainage model *Manuscript*.

In this online version of the thesis, the papers are not included but can be obtained from electronic article databases e.g. via www.orbit.dtu.dk or on request from: DTU Environment, Technical University of Denmark, Miljøvej, Building 113, 2800 Kgs. Lyngby, Denmark. reception@env.dtu.dk.

In addition, the following publications, not included in this thesis, were also published during the PhD study:

Andersen, S. T., Erichsen, A. C., Mark, O. and Albrechtsen, H.-J. (2011) *Gastroenteritis: A waterborne outbreak affected 430 people in Copenhagen harbor during ironman competition –Could this be avoided?* Proceedings from the FIPS 2011 conference; "Feacal indicators: problem or solution? Has technical progress reduced the need for faecal indicators?", Edinburgh Conference centre, UK, 6th -8th June 2011, p. 25. Awarded: Best poster presentation. Poster presentation.

Andersen, S. T., Erichsen, A. C., Mark, O. and Albrechtsen, H.-J (2012). *Effects of an extreme rain event: a quantitative microbial risk assessment of a case of polluted bathing water in Copenhagen, Denmark*. Presentation at the International Conference; "Water Sensitive Cities" arranged by The Society of Danish Engineers (IDA), Copenhagen, 28th August 2012. Oral presentation.

Sharma, A. K., Lyngard-Jensen, A., Vezzaro, L., **Andersen, S. T.**, Brodersen, E., Eisum, N. H., Jacobsen, B. S., Høgh, J., Gadegaard, T. N., Mikkelsen, P. S., Rasmussen, M. R. (2014) *Advanced monitoring of Combined Sewer Overflows: What to measure and how to measure*. ICUD 2014, the 13th International Conference on Urban Drainage, Sarawak, Malaysia, 7-12 September 2014. Contribution to conference article.

Andersen, S. T., Mark, O., Albrechtsen, H-J. (2014). *Quantitative microbial risk assessments of the impacts of flooded basements in urban areas by combining quantitative microbial data with hydrological software*. Proceedings from the IWA 2014, World water Congress and Exhibition, Lisbon, Portugal, 21th-26th September 2014. Oral presentation.

Finally, the PhD project resulted in seminar contributions and contribution to a Danish journal paper in connection to a century rainfall July 2011, in the central of Copenhagen, Denmark, causing major flooding.

Clauson-Kaas, J, Kjerulf, A., Holt, J., **Andersen, S. T.**, Albrechtsen, H-J (2011). Skybrud, Sundhed og Ulykker. *DanskVand*, Vol. 79 (6), p. 36-38.

December 2014

Signe Tanja Andersen

Acknowledgements

I wish to deeply thank my two supervisors, for sharing their knowledge and experience and for their help and support for better and for worse. In particular, I would like to thank Hans-Jørgen Albrechtsen for giving me the opportunity to do the PhD study and guide me through the entire process and Ole Mark for introducing me to the world of water quality modelling and give feedback on any other aspects of the PhD work. Also a special thanks to Òluva Vang for sparring with me, scientifically and when I needed to talk.

A special thanks to my office mates at DTU Environment, Berit, Suzi, Luca, Elham, Sara, Vianney. Also a thanks to my colleagues from the research group Water Supply, including Sarah, Carson, Florian, Mathilde, Karolina, Camilla, Ravi, and Charlotte, as well as Hjalthe and Ida. Also a great thanks to the support people at DTU environment, including Rene, Lene, Sabrina, Bent, Anne, Lisbeth, Torben and many more.

Thanks to colleagues from the SWI project including Anitha, Morten, Luca, Peter. Also thanks to the collaborators Carsten Thirsing and his staff from Biofoss, Kasper Juel-Berg and Jesper Thyme from HOFOR, Århus Vand, and Hvidovre Vandforsyning for helping with sampling and analysis.

It has been a pleasure to periodically working at DHI, together with the experienced staff, especially Claus Jørgensen and Anders Erichsen, and at DTU food together with Anna Charlotte Schultz and Resadije Idrizi.

Last, innermost thanks to my family and friends for encouragement and support that helped me complete this work. Special warm thanks to my husband Anders for fully supporting me especially when I was feeling worn out and for bearing with me when I was an absent-minded wife. Thanks to my parents and parents-in-law for helping with taking care of my two boys Jonathan and Anton whenever needed and for support throughout the whole PhD study, even so they are still wondering what exactly I was doing for the last five years.

Og på dansk vil jeg sige: “Mange tak for hjælpen til alle som har hjulpet mig i årenes løb, i er årsagen til at jeg fik gennemført!”.

Summary

Extreme rainfall overloads combined sewers, thereby causing flooding in urban areas, and if the public is exposed to flooding, they are at risk of acquiring gastrointestinal diseases. This is a known problem and is expected to increase because the frequency and intensity of extreme rainfall are expected to increase in the future. To ensure public health during extreme rainfall, solutions are needed, but limited knowledge on microbial water quality, and related health risks, makes it difficult to implement microbial risk analysis as a part of the basis for decision making.

The main aim of this PhD thesis is to identify the limitations and possibilities for optimising microbial risk assessments of urban flooding through more evidence-based solutions, including quantitative microbial data and hydrodynamic water quality models. The focus falls especially on the problem of data needs and the causes of variations in the data.

Essential limiting factors of urban flooding QMRAs were identified as uncertainty regarding ingestion volumes, the limited use of dose-response models and low numbers of microbial parameters measurements and absent validation of the risk assessments. Because improving knowledge of ingestion volumes and of dose-response models involves many difficulties, including ethical considerations, they are therefore less manageable than measuring microbial parameters. But when aiming at predicting the risk of infection from exposure to urban floodwater, measurements from flooding episodes are difficult to acquire, and so it is actually more feasible to use dynamic water quality models, which has the advantage of including dilution, spatial distribution and changes over time. However, determining inputs into water quality models is a challenge, because wastewater composition varies greatly according to location, as demonstrated for microbial, chemical and physical parameters between sub-catchments.

Variations in wastewater quality, as well as rainfall, system and environmental effects (solar radiation), cause knock-on variations in floodwater composition, because they are functions of these parameters. Variations between locations have been demonstrated through measurements of microbial concentrations in flooding episodes, while changes in microbial concentrations over time have been demonstrated through a survival and decay study, where decay was substantial in the presence of UV light (potential sunlight) and varied according to turbidity and depth.

Risk analysis could be integrated for use in risk management more than they are today, which involves the use of measurements along with hydrodynamic water quality models. In this PhD thesis two risk assessments were conducted with employment of hydrodynamic water quality models. One risk assessment examined contaminated bathing water, and here dilution and transport were modelled through a drainage model and a hydrodynamic bathing water model. The maximum concentrations of pathogens in wastewater found in the literature were used to estimate risk, and by comparing the model results with an epidemiological study of the same event, the concept of using hydrological models to estimate water quality – and thereby estimate risk – was improved. Another urban flooding risk assessment used average measured concentrations of pathogens in wastewater as inputs into a drainage model, to estimate the pathogen concentration in the floodwater. Compared to the risk assessment for the contaminated bathing water, the concept was improved, because actual pathogen measurements rather than literature-based values were used as inputs into the drainage model. Furthermore, the drainage model was validated by comparison of modelled and measured microbial concentrations in CSOs. The model result was used in the analysis of the risk of infection from exposure to urban flooding which resulted in a risk of 10^{-3} to 10^{-1} from exposure to flooding, both from cleaning up flooding, but also when wading through a flooded area.

The results in this thesis have brought microbial risk assessments one step closer to more uniform and repeatable risk analysis by using actual and relevant measured data and hydrodynamic water quality models to estimate the risk from flooding caused by overloaded combined sewers. This approach is useful in supporting decision making regarding optimising sewer systems in the future to ensure public health.

Dansk sammenfatning

Ekstreme regnhændelser kan føre til overfyldte kloaker som kan oversvømme byområder. Hvis befolkningen udsættes for kontakt til oversvømmelsesvand, er der risiko for at få gastrointestinale sygdomme. Dette er i dag et kendt problem som forventes at tiltage yderligere, da ekstreme regnhændelser forventes at forekomme hyppigere og med højere intensitet i fremtiden. Der er brug for løsninger til at sikre befolkningens sundhed under disse ekstreme regnhændelser, men den begrænsede viden om den mikrobielle vandkvalitet i oversvømmelser, og den dertilhørende sundhedsrisiko, gør det svært at implementere mikrobielle risikovurderinger som et redskab til beslutninger.

Hovedformålet med PhD-afhandlingen er at identificere begrænsningerne og mulighederne for optimering af mikrobielle risikovurderinger for urbane oversvømmelser, ved mere evidens baserede løsninger, der inkluderer kvantitative data og hydrodynamiske vandkvalitetsmodeller. Der er særligt fokus på problemstillingerne om databehov og årsager til variationer i vandkvalitet.

De væsentligste begrænsende faktorer for kvantitative mikrobielle risiko analyser for urbane oversvømmelser blev identificeret som usikkerheder for indtag ved kontakt med oversvømmelsesvand, den begrænsede brugbarhed af dosis-responsmodeller, de få målinger for mikrobielle parametre, samt den manglende validering af risikovurderingen. At øge vores viden om indtag samt forbedre dosis-responsmodeller er væsentligt vanskeligere end at måle for mikrobielle parametre, da det inkluderer etiske aspekter. Men når målet er, at forudsige risikoen for infektion fra kontakt til urbane oversvømmelser, kan målinger af mikrobielle parametre være svære at skaffe, hvorved det er mere brugbart at anvende dynamiske vandkvalitetsmodeller. De har desuden den fordel, at de inkluderer viden om fortynding, fordeling over land samt ændringer over tid. Men det er en udfordring at bestemme inputs til disse vandkvalitetsmodeller fordi spildevandssammensætningen varierer meget i forhold til lokalitet, hvilket var demonstreret for forskellige oplande, og mikrobielle, fysiske og kemiske parametre.

Sammensætning af oversvømmelsesvand varierer ligeledes i høj grad afhængig af tid og sted som resultat af spildevandskvalitet, regnhændelse og ydre påvirkninger (for eksempel sollys). Variationer imellem lokaliteter blev påvist i målinger af de mikrobielle koncentrationer i oversvømmelsesvand. Variationen over tid blev påvist i et henfaldsstudie, hvor henfaldet var bety-

deligt i UV-lys (svarende til sollys). Henfaldsraten afhang ydermere af turbiditet og dybde.

Fremover kan risikoanalyser, i højere grad end de er i dag, integreres i risikohåndtering, fx med anvendelse af hydrodynamiske vandkvalitetsmodeller. Denne PhD afhandling omfatter to risikovurderinger med hydrauliske vandkvalitetsmodeller. I den ene risikovurdering, for forurenet badevand, blev fortynding og transport modelleret med en afløbsmodel og en hydrodynamisk badevandsmodel. Maksimale litteraturværdier for koncentration af patogener i spildevand blev anvendt i risikoanalysen. En sammenligning af modelresultaterne, og et epidemiologisk studie af samme hændelse, viste det hensigtsmæssige i konceptet med hydrologiske modeller til at estimere mikrobiel vandkvalitet, og dermed risiko. I en anden risikoanalyse af oversvømmelse i byområder, under en ekstrem regnhændelse, anvendtes gennemsnitskoncentrationer af målte mikrobielle parametre i spildevand, som input til en afløbsmodel. Sammenlignet med risikoanalysen for forurenet badevand er konceptet dermed forbedret ved at anvende målinger i stedet for litteraturværdier, som input i afløbsmodellen, og afløbsmodellen blev valideret ved sammenligning af modelresultater og målinger i overløb for forskellige regnhændelser. Modelresultaterne blev anvendt i en risikoanalyse for risikoen for infektion ved kontakt med oversvømmelsesvand. Analysen viste en væsentlig risiko for mikrobiel infektion på 10^{-3} til 10^{-1} ved eksponering til oversvømmelsesvand, både ved oprydningsarbejde, men også ved at passere igennem oversvømmede områder

Resultaterne i denne afhandling har bragt kvantitative mikrobielle risikoanalyser et skridt nærmere mere ensartethed og repeterbarhed ved at anvende målte data og hydrodynamiske vandkvalitetsmodeller til at estimere risikoen ved oversvømmelser forårsaget af overfyldte kloaker. Dette værktøj kan anvendes i beslutninger omkring optimering af kloaksystemet for at sikre den offentlige sundhed.

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1 Introduction

Heavy rains are predicted to occur more frequently and with greater intensity because of climate change (Hartmann et al. 2013), and a subsequent increase in urbanisation is also expected (Biggs et al. 2011). Together, these factors will increase the load on sewers and cause more flooding, although sewers already frequently overflow in cities worldwide (Islam et al. 2010, WHO 2013).

Sewer flooding causes huge infrastructural problems, damage, accidents and diseases aligned with the very real possibility of mortality. Disease outbreaks caused by pathogenic microorganisms in urban flooding are a well-recognised problem (Ahern et al. 2005, Veldhuis et al. 2010, Fewtrell et al. 2011, Aldermann et al. 2012, Cann et al. 2013), and these microbial hazards vary according to geographical location; for example the bacteria, *Vibrio spp*. and *Leptospira*, two major causes of gastrointestinal disease, are found in regions with low sanitation, such as Asia and Africa, whereas Norovirus, Cryptosporidium and *Campylobacter spp.* are major causes of gastrointestinal disease in regions with high sanitation, such as northern Europe (Cann et al. 2013).

The different approaches taken to protect citizens against flooding are limited by environmental legislation, demographic changes and urban development (Biggs et. al 2011). Implementing solutions to overcome flooding, such as redesigning sewer systems, making structural changes through the construction of basins and optimisation, are very expensive. When choosing between different solutions the decision should be based on the best possible information. Risks to human health through exposure to microbial hazards following flooding episodes should therefore be identified clearly, considered and evaluated, in order to act as the basis for decision making and implementing solutions related to public risk.

Quantitative microbial risk assessment (QMRA) is a tool that combines information on pathogenic microorganisms with mathematical models, in order to describe transmission through environmental exposure and to define human health risks (Sterk et al. 2013). QMRA can be used to estimate or evaluate the risk of infection from e.g. rainfall events. To carry out a QMRA, microbial data are needed, but today such data are limited because of difficulties in sampling floodwater due to it’s stochastic nature, including limited accessibility and large variations in microbial concentrations caused by the given situation. Furthermore, when sampling floodwater, the results depend on rainfall and the system, which are not representative of the whole flooding event. Solutions other than sampling floodwater for QMRA purposes are therefore required.

1.1 PhD thesis approach

The approach used for applying measurements and hydrodynamic water quality models in an urban flooding QMRA are presented in Figure 1. To estimate the risk of infection through exposure to floodwater, microbial concentrations in wastewater under dry weather conditions, and time series of rainfall, are used as inputs into a drainage model (Mike Urban). Microbial concentrations represent background concentrations in combined sewers, before rainfall runoff enters the system. The model can then estimate the initial microbial concentrations in floodwater and concentrations of microorganisms on exposure are estimated by use of decay rate constants, reflecting the flooding scenario. These microbial concentrations and ingestion volumes are then used as inputs into selected dose-response models to find the risk of infection.

This approach requires a range of validation steps. First, the output of the drainage model can be validated through the use of measured microbial concentrations in combined sewer overflows, in order to ascertain if dispersion and advection data are correct in the model. Drainage model output can then be validated through by measured microbial concentrations in flooding events. The estimated risk of infection can be validated by comparison to the risk of infection estimated from measured microbial concentrations in flooding events or to reported illnesses from epidemiological studies.

In this PhD thesis the limitations and possibilities for QMRA improvements are identified and discussed with a focus on the collection of data relating to microorganisms in combined sewers and floodwater.

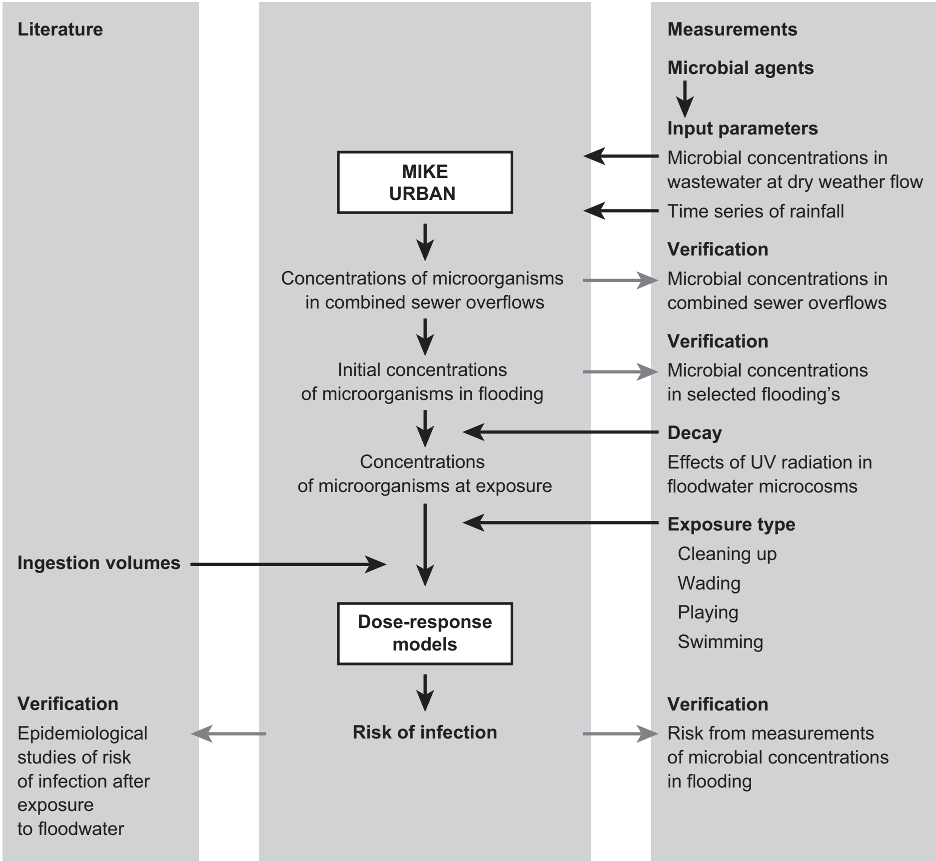

Figure 1. Flow diagram for the approach of microbial risk assessment using a combination of measurements and hydrodynamic models (MIKE URBAN) from input to output, risk characterisation and verification (Andersen et al. IV).

1.2 Objectives and aims

The main aim of this PhD thesis is to identify the limitations and possibilities for optimising microbial risk assessments of urban flooding through more evidence-based solutions, including quantitative microbial data and a hydrodynamic water quality model. The focus falls especially on the problem of data needs and the causes of variations.

This PhD thesis consists of an introduction and four papers. The main objectives are to identify possibilities for improving QMRA, in relation to health risks, in association with urban flooding. Microbial data from wastewater, combined sewer overflows and flooding events are collected for use as input data and to validate a hydrodynamic water quality model, to develop more precise and repeatable risk assessments of urban flooding.

The aims of the PhD thesis are:

1. *To place quantitative microbial risk assessments in the context of urban flooding and to determine the limitations of microbial risk assessments of urban flooding episodes.*
2. *To identify the possibilities for improving quantitative microbial risk assessments of urban flooding episodes and to evaluate the applicability of the outcome.*

The aims of the PhD thesis will be achieved through the partial aims defined in the four papers included in the thesis, namely:

- To develop a strategy to collect quantitative microbial data from sewers and to characterise variations in microbial concentrations in wastewater during dry weather, which are useful as inputs into water quality modelling (Paper I)
- To investigate the effects of UV radiation on the inactivation of microorganisms in floodwater and to determine inactivation in the dark, in order to evaluate the importance of inactivation in floodwater (Paper II)
- To investigate if the approach of combining microbial concentration data with hydrodynamic models is useful as an input for microbial risk assessments (Paper III)
- To validate the hydrodynamic model by comparing modelled and measured microbial concentrations in combined sewer overflows (Paper IV)
- To apply the outputs of the validated hydrodynamic model in a quantitative microbial risk assessment and then evaluate the microbial risk assessment by using floodwater measurements (Paper IV)

2 System description and water quality variations

During heavy rainfall, combined sewers can become overloaded by rainfall runoff, thus causing urban flooding. Urban floodwater is contaminated with microbial pathogens, and exposed persons may contract microbial diseases. To assure the safety and health of the population, existing urban flooding conditions should be understood, which requires knowing what factors determine risk. System construction (combined sewers, separate sewers) and rainfall determine the risk of exposure and the type of flooding, while wastewater composition determines what runs through the pipes according to different variations such as temporal, seasonal and catchment. Environmental conditions, such as solar radiation, determine floodwater quality over time. The interaction between these systems during rainfall and the following floodwater quality can be estimated by hydrodynamic water quality models. To identify when exposure occurs, and the relevance of parameters such as time (day, night and day of the week), length, and type of flooding should be considered. The conditions identified as the most important to consider when carrying out a risk analysis of urban sewer flooding are therefore:

- The structure of urban sewer systems
- Wastewater composition and variations
- Quantity and duration of rainfall
- Variations in floodwater quality
- Data needs for modelling floodwater quality

2.1 Structure of urban sewer systems

The purpose of sewer systems is to collect and transport wastewater away from urban areas, in order to ensure a high level of public hygiene and the low occurrence of infectious diseases. Generally, there are two sewer systems: 1) combined sewers, where wastewater and stormwater are collected and transported through one system of pipes to a wastewater treatment plant, and 2) separate sewers, where wastewater and stormwater are collected and transported through two separate pipe systems to a wastewater treatment plants or other recipient. The distribution of sewers in Denmark is close to 50% combined sewers and 50% separate sewers, while these figures are closer to 70% combined sewers and 30% separate sewers in France, Germany and the UK, of which the most elderly urban areas have typically combined sewers, while more recent urban areas have separate systems (Naturstyrelsen 2014, Butler & Davies 2011).

During heavy rains, combined sewers may become overloaded, thereby leading to combined sewer overflows (CSOs) into nearby ‘recipients’ such as harbours, rivers or streams. This is undesirable because CSOs may cause eutrophication and oxygen depletion and contaminate bathing water (Figure 2). During extreme rainfall events, CSOs may not be sufficient to alleviate the load of sewers, and so flooding of houses, streets and other areas may occur (Figure 2). As a result, citizens may be exposed when playing, wading, swimming or cleaning up flood damage, which is a problem because urban floodwater can be home to high numbers of pathogenic microorganisms (Alderman et al. 2012, Smith et al. 2007), thus putting people at risk of getting microbial infections and illnesses. Existing combined sewers could be replaced with separate sewers, but this would be a difficult and costly undertaking in high-density cities, and thus more technical solutions are needed.

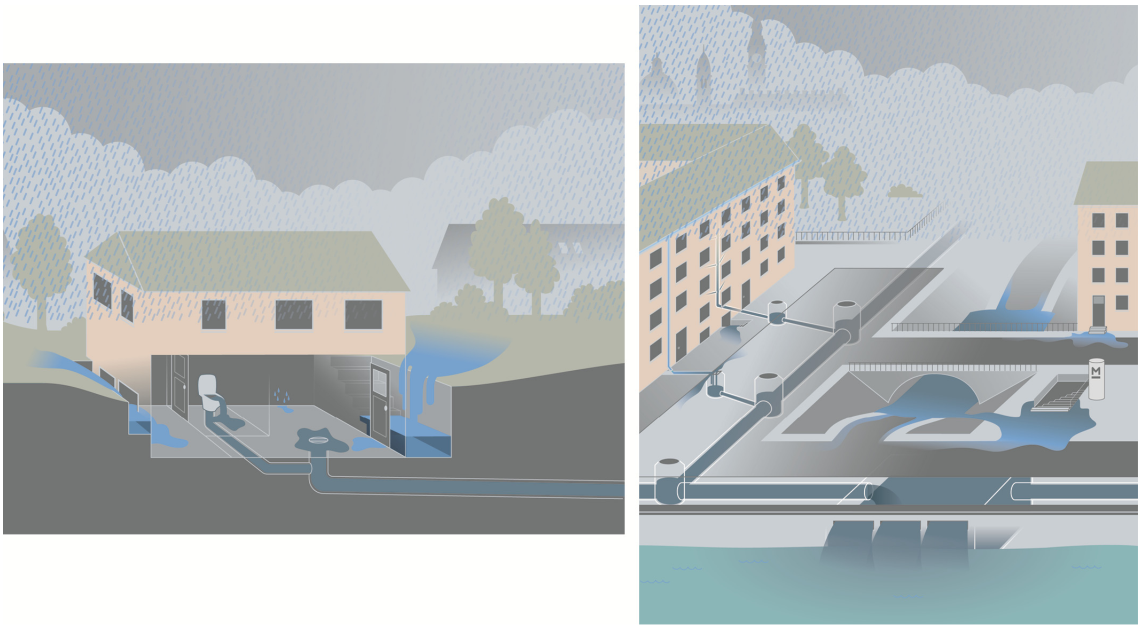

Figure 2. The basic elements of combined sewer systems. Rainfall runoff from roofs and streets run into combined sewers, causing combined sewer overflows and flooding through manholes in low-lying areas, viaducts and streets and through toilets and gullies (Illustration: Tobias Scheel Mikkelsen, colours modified).

2.2 Wastewater composition and variations

In order to model microbial water quality during flooding, knowledge of microbial concentrations and variations in wastewater at dry weather flow is required.

2.2.1 Microbial concentrations

The major objective of urban drainage systems is to protect human health, especially from exposure to human faeces. Nonetheless, microorganisms are not routinely monitored, and when they are monitored it is typically microbial indicators that are examined whereas disease-causing pathogens are less often observed (Butler & Davies 2011). However, indicators and pathogens are present in high and varying concentrations in wastewater (Table 1), and these concentrations vary according to differences in water consumption and the type of water use, infiltration and exfiltration (Henze et al. 2006, Butler & Davies 2011).

The presence of pathogens in wastewater depends on infections in populations, which can see temporal and seasonal variations (Smith et al. 2007). Consequently, in wastewater indicator bacteria such as *E. coli* and *Enterococci* and pathogens such as *Campylobacter* and Norovirus are monitored over several log levels (Table 1).

Table 1. Examples of measured concentrations of indicators and pathogenic microorganisms in wastewater*

Indicators	**Concentration/100mL***
Escherichia coli	10^3-10^7
Enterococci spp.	10^2-10^6
Pathogens	
Campylobacter spp.	$10^1 – 10^6$
Salmonella spp.	$>1 – 10^6$
Norovirus	>1-10^8
Rotavirus	$4*10^2 – 8.5*10^4$
Cryptosporidium (oocysts)	$>10^0 – 0.4*10^2$
Giardia (cysts)	$10^1 – 2.0*10^4$

*Andersen et al. I, Andersen et al. III, Lodder and de Roda Husman 2005, Haramoto et al. 2006, Katayma et al. 2008, Kay et al. 2008, Soonthormonda 2008, Kim et al. 2009, Rijal et al. 2009, Veldhuis et al. 2010, WHO 2003.

2.2.2 Diurnal variations

Typically, any increase in water use is associated with domestic activities, e.g. toilet use and showering, and as a result flow increases in the morning and evening hours (Figure 3) (Butler et al. 1991, Pantsar-Kallio et al. 1999). Toilet use contributes the most to wastewater, and adults produce 200-300g of faeces per day and the bulk of microorganisms in wastewater are from faeces, whereas urine is relatively free from microbes (Butler & Davies 2011).

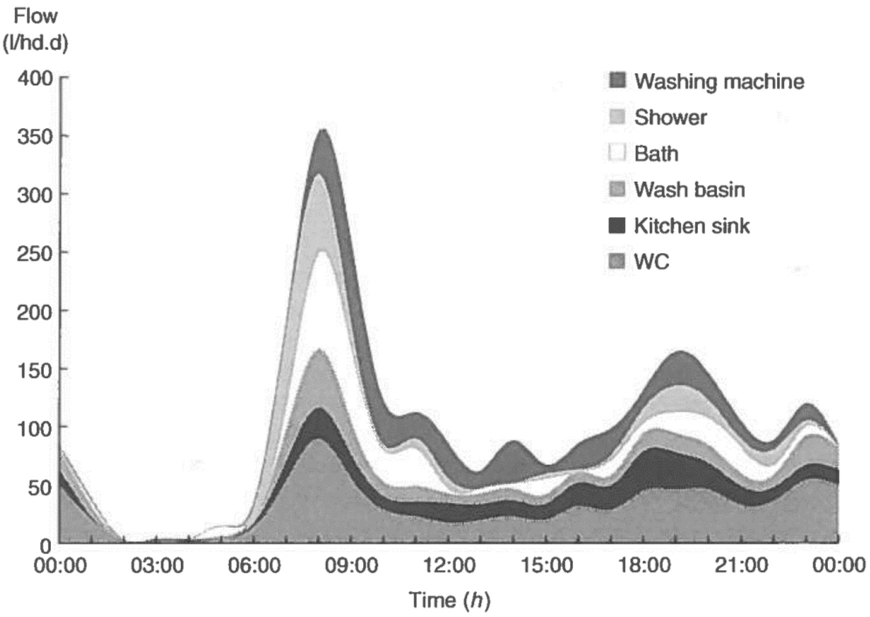

Figure 3. Diurnal discharge patterns (flow) of domestic wastewater during dry weather conditions, with peaks in flow in the morning (6-10 am) and evening hours (18-21 pm) (Butler & Davies 2011).

The diurnal patterns of microorganisms and wastewater quality parameters such as ammonia and suspended solids were examined during dry weather conditions. The microbial concentrations increase in morning hours (4-12 am) and evening hours (18-22 pm) downstream three urban sub-catchments, located in Denmark (Figure 4). The diurnal patterns of microbial concentrations change according to the retention time of the catchments, and inlet to the receiving wastewater treatment plant the diurnal pattern is delayed as compared to the diurnal pattern of the upstream sub-catchments, which demonstrates the importance of retention time and thereby wastewater quality (Andersen et al. I). Suspended solids and ammonia follow a similar diurnal pattern in residential catchments, although without being reflected by microbial parameters (Gaspari et al. 2008, Lamprea et al. 2011), while in another study investigating microbial parameters at the inlets to two wastewater treatment plants, the diurnal patterns of ammonia, total suspended solids and indicator bacteria *E. coli* and *Enterococci* were based on retention time (Lucas et al. 2014). Indicator bacteria and ammonia follow a similar diurnal pattern, probably because indicator bacteria are associated with faecal matter (Lucas et al. 2014). It is therefore also likely that they are transported similarly in sewers.

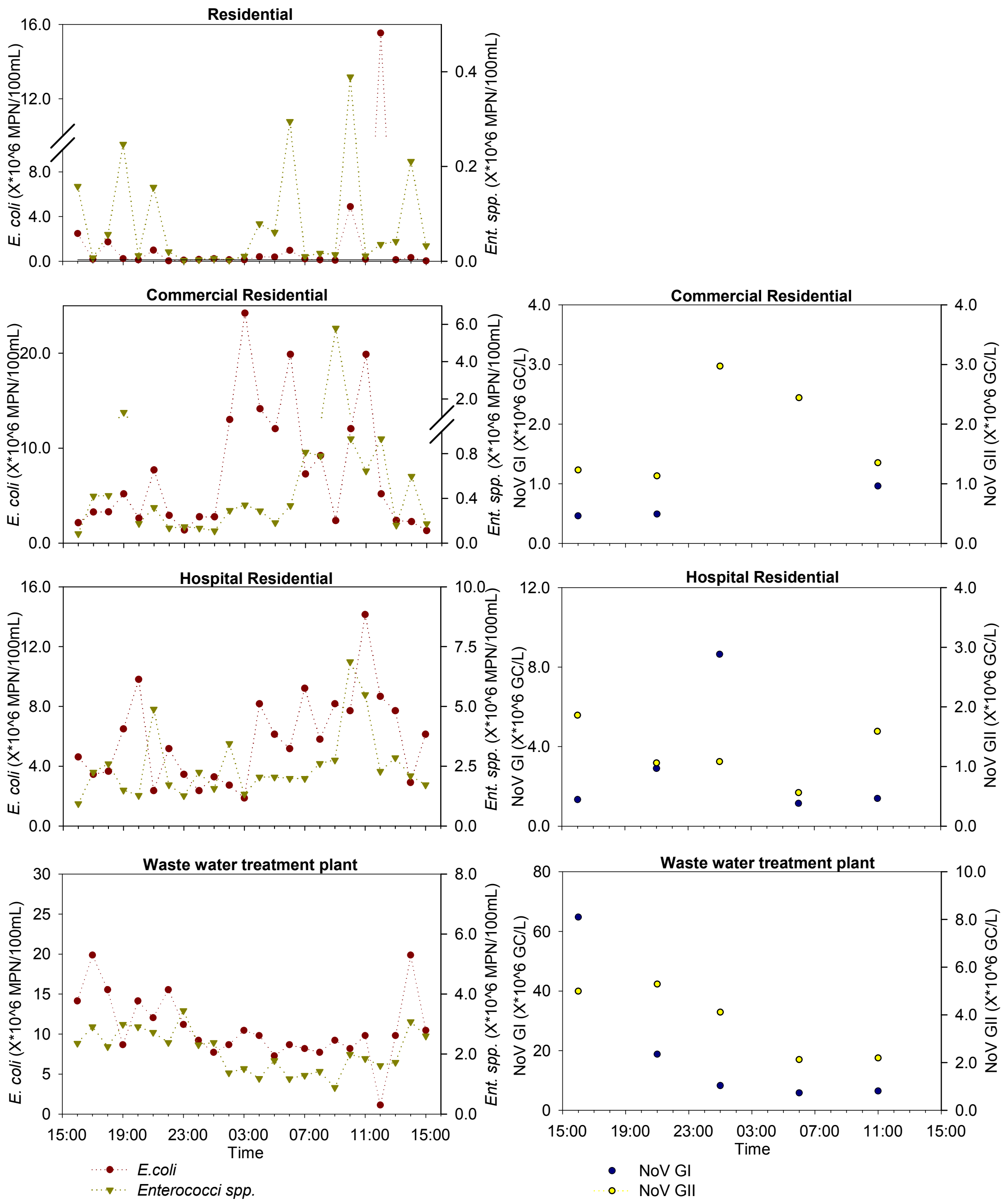

Figure 4. Diurnal variations of *E. coli*, *Enterococci* and Norovirus in wastewater in dry weather conditions for three sub-catchments; Residential, Commercial-Residential, Hospital-Residential, and an inlet into a wastewater treatment plant (Andersen et al. I).

The diurnal pattern of microorganisms in wastewater is rarely investigated, and only few studies are published on the topic. Diurnal patterns for the indicator bacteria were comparable to one study where concentrations increased during the morning and reached maximum levels at night (Madoux-Humery et al. 2013), and to another study where the concentration was lower between midnight and 10 am than between 12 pm and 10 pm (Lucas et al. 2014). The diurnal patterns were less similar to what were found in a single residential catchment in Japan, where concentrations were moderately constant from noon to midnight, decreased during the night and then increased in the morning hours (7-8 am) (Figure 5) (Kim et al. 2009).

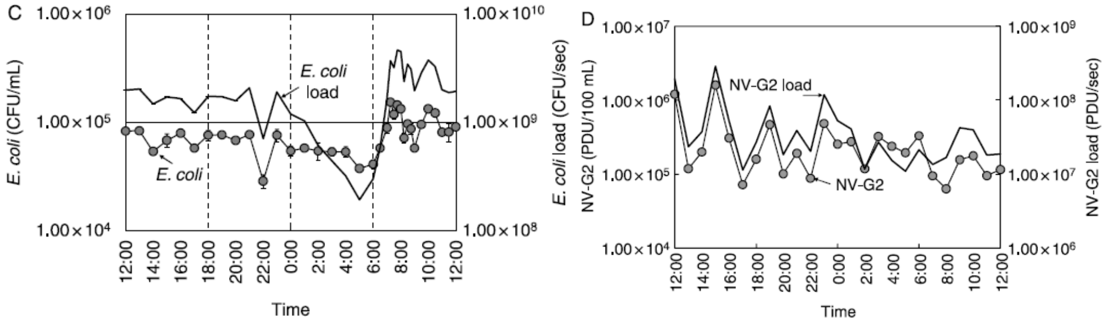

Figure 5. Diurnal concentration and load time-series variations of *E. coli* and Norovirus GI and GII in dry weather conditions at a single residential catchment (Kim et al. 2009).

This and the study by Andersen et al. (I) were conducted for urban residential catchments, which were similar in size with sampling downstream the catchments, so retention times are assumed to be similar for the catchments. Differences in the diurnal patterns between the two studies may therefore reflect varying cycles of the populations or different social activity in the community, which demonstrates that generalising diurnal patterns is challenging and requires knowledge of land use and the type of society and quantitative data.

In conclusion, microbial concentrations vary according to the diurnal pattern, but the diurnal pattern vary according to the retention time in the catchment

2.2.3 Catchment variations

Catchment variations were examined through the simultaneous downstream sampling of three urban sub-catchments and at an inlet to a receiving wastewater treatment plant (WWTP) in a catchment area in Denmark. The results show significant differences between concentrations of *E. coli*, *Enterococci* and Norovirus between the three sub-catchments and at the inlet to the WWTP (p<0.05) (Figure 6), with up to two log folds of difference in microbial concentrations. Additionally, high standard deviations are present, thereby suggesting that estimating wastewater quality during dry weather conditions requires representative sampling as well as the presence of temporal variability (Andersen et al. I). Most importantly, catchment variations were higher than diurnal variations, and hence they should be considered, rather than diurnal variations, when aiming at characterising wastewater quality.

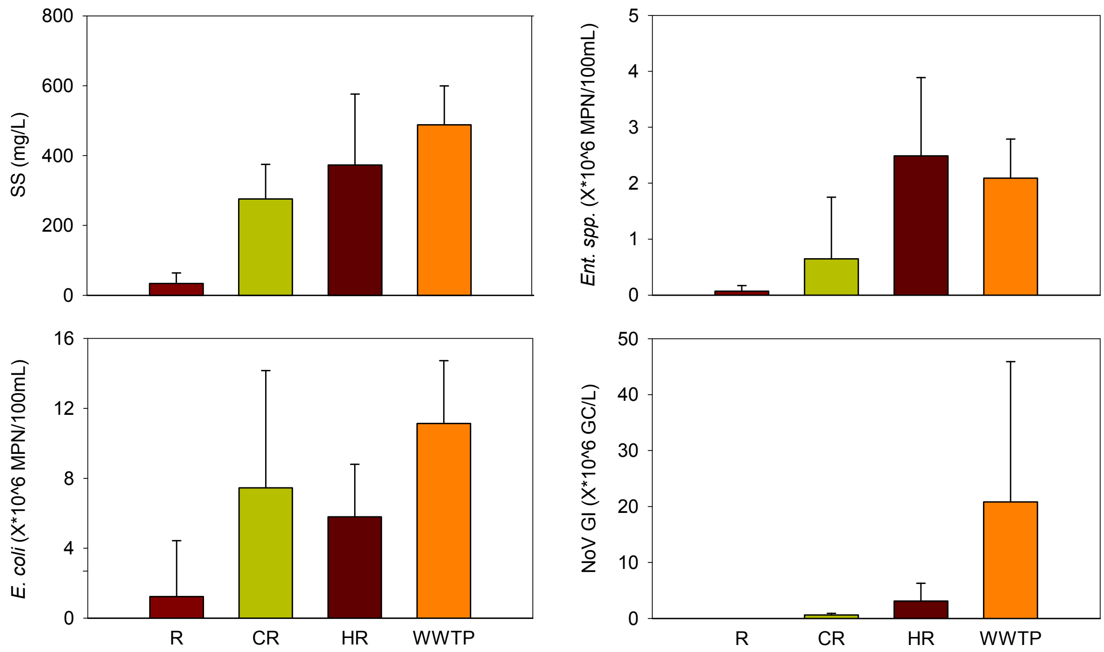

Figure 6. Concentrations of suspended solids, *E. coli*, *Enterococci spp.* and Norovirus GI (NoV), (± standard deviations) at different sub-catchments; Residential (R), Commercial-Residential (CR), Hospital-Residential (HR) and wastewater treatment plant (WWTP) inlet (Andersen et al. I).

To our knowledge only a few studies have investigated differences between catchments relating to microbial wastewater during dry weather conditions. One study investigated concentrations of microbial indicators entering two wastewater treatment plants with different land uses and sewer combinations, and they found significant differences (Lucas et al. 2014). Also, in a study of two catchments but with different land uses (mainly residential and institutional), the authors also found differences in microbial concentrations (Madoux-Humery et al. 2013). Land use is therefore a very important factor when determining microbial concentrations in wastewater.

The catchment variations were strongest in the small residential catchment (1.9 ha), followed by the commercial-residential catchment (54 ha) and the hospital-residential catchment (163 ha), as well as the inlet into the WWTP (4700 ha) (Andersen et al. I). Low variations at the inlet into the WWTP are probably the result of the mixing wastewater, as found previously for *E. coli* and *Enterococci* in wastewater (Rijal et al. 2009, Sterk et al. 2008). Typically, variations in microbial parameters in larger catchments (>50 ha) are >10- to 30-fold (Andersen et al. I, Kim et al. 2009, Sterk et al. 2008, Veldhuis 2010). In smaller catchments (<50 ha) there is >30-fold variation (Andersen et al. I). The spatial scale tends to smoothen out local effects and homogenise the quality of wastewater (Lucas et al. 2014), which ideally should equate to less variation. This explains the lower variations found in larger catchments and at wastewater treatment plant inlets.

In conclusion, catchment variations where significant and certainly stronger than diurnal variations. Moreover, catchment size determines variations in wastewater quality.

2.3 Variations in floodwater quality

When estimating the risk of infection from exposure to flooding, the results should be validated. Therefore, when using water quality models for estimating microbial concentrations in floodwater, measurements are required, to verify the water quality model and are valuable for explaining the causes of differences between modelled and measured microbial concentrations.

2.3.1 Pathogens in floodwater

Sanitation levels and general diseases in the population are two major parameters influencing the presence of pathogens in flooding events (Alderman et al. 2012). Different pathogens are therefore detected in varying concentrations in sewers impacted by urban flooding or associated outbreaks (Table 2).

From investigations of the microbial concentrations in floodwater the indicator bacteria *E. coli* and *Enterococci* were detected in substantial concentrations in seven urban floodwater samples, whereas Norovirus was only detected at one single location (prevalence 14%). When monitoring CSOs the trend was similar, and indicator bacteria were detected in every sample, whereas Norovirus was detected with a prevalence of 20-100%, depending on the event. *Campylobacter* was detected in 100% of the samples from CSOs (Andersen et al. IV). In a study of urban flooding in the Netherlands, the prevalence was 61% for *Campylobacter jejuni*, 35% for Giardia, 30% for Cryptosporidium and 29% for Norovirus (Man et al. 2014). Another study in the Netherlands found *E. coli*, *Enterococci* and *Campylobacter* in all investigated samples (Veldhuis et al. 2010). In the United Kingdom the prevalence for Giardia and Cryptosporidium is 10-20% (Fewtrell et al. 2011). Bacterial indicators are thus frequently detected in urban floodwater, but the presence of these indicators does not equate to the presence of pathogens, which are irregularly detected in flooding (Table 2).

Table 2. Selected publications of pathogens and their concentrations in CSOs, floodwater, or resulting in outbreaks after a flooding.

Organism	Country	Water Source	Concentrations*	References
Campylobacter spp.	DE	Sewer flooding	10^6 CFU/L	Rechtenburg et al. 2009
	NL	Sewer overflow	Detected	Veldhuis et al. 2010
	DK	CSO	$2\text{-}3.5*10^5$ CFU/L	Andersen et al. IV
C. jejuni	NL	Sewer flooding	7-1,500 MPN/L	Man et al. 2014
Salmonella	UK	River flooding with sewage input	2-1,3900/L	Fewtrell et al. 2011
Leptospira	IN	Residence in flood risk region with open sewers	N.M.	Pappachan et al. 2004
	BR	Residence in flood risk region with open sewers	N.M.	Reis et al. 2008
	DK	Sewer flooding	N.M.	SSI, 2011
Vibrio Cholera	BD	River and stream with sewage input	N.M.	Hashizume et al. 2008
Cryptosporidium	UK	River flooding with sewage input	0.1-1/L	Fewtrell et al. 2011
	NL	Sewer flooding	0.1-10 oocysts/L	Man et al. 2014
Giardia	UK	River flooding with sewage input	0.1-32/L	Fewtrell et al. 2011
	NL	Sewer flooding	0.1-142 cysts/L	Man et al. 2014
Norovirus	AT	Sewer flooding	N.M.	Schmid et al. 2005
	US	Hurricane Katrina	N.M.	Jablecki et al. 2005
Norovirus GI	ID	River flooding with sewage input	200 PDU/L	Phanuwan et al. 2006
	NL	Sewer flooding	$6.1\text{-}33*10^2$ PDU/L	Man et al. 2014
	DK	Sewer flooding	$8.1*10^4$ GC/L	Andersen et al. IV
		CSO	$1.1\text{-}14.1*10^4$ GC/L	Andersen et al. IV
Norovirus GII	NL	Sewer flooding	530-40,000 PDU/L	Man et al. 2014
	DK	Sewer flooding	$3*10^6$ GC/L	Andersen et al. IV
		CSO	$3*10^5\text{-}1*10^6$ GC/L	Andersen et al. IV
	ID	River flooding with sewage input	$2.2\text{-}3*10^4$ PDU/L	Phanuwan et al. 2006
Enterovirus	ID	River flooding with sewage input	$12\text{-}72*10^3$ PDU/L	Phanuwan et al. 2006
	NL	Sewer flooding	$1.6\text{-}40*10^3$ PDU/L	Man et al. 2014
Hepatitis A	ID	River flooding with sewage input	$7.1\text{-}8.7*10^3$ PDU/L	Phanuwan et al. 2006
Adenovirus	ID	River flooding with sewage input	$51\text{-}60*10^3$ PDU/L	Phanuwan et al. 2006

*N.M.; Not measured

2.3.2 Changes in microbial concentrations in sewers according to rainfall runoff into sewers

Concentrations of microorganisms in rainfall runoff (stormwater) were measured as 1-261 MPN/100 mL *E. coli*, 65-1,046 MPN/100 mL *Enterococci* and either >1 per CFU/100 mL or 4,00-18,000 CFU/100 mL *Campylobacter* (Andersen et al. IV). Concentrations of indicators and *Campylobacter* in rainfall runoff were therefore several log folds lower than in wastewater. In another study of microbial concentrations in rainfall runoff the concentration of *Campylobacter* was <1-62 MPN/ 100 mL, and pathogens such as Giardia, Cryptosporidium and Norovirus went undetected in both rainfall runoff and storm sewers (Man et al. 2014). Dilution by rainfall runoff in combined sewers is therefore assumed to lower microbial concentrations.

Microbial concentrations are more than ten times lower in CSOs than in wastewater (Figure 7), and generally microbial concentrations are lower in CSOs than in wastewater (Andersen et al. IV, Madoux-Humery et al. 2013, McCarthy et al. 2009). Microbial concentrations in flooding are consequently lower than in CSOs because of the level of dilution by the less contaminated rainfall runoff (Figure 7) (Andersen et al. IV).

In conclusion, the resulting microbial concentrations, from the lowest to the highest, are therefore: wastewater → combined sewer overflow → flooding → rainfall runoff, whereby microbial contaminations in urban flooding originate mainly from wastewater.

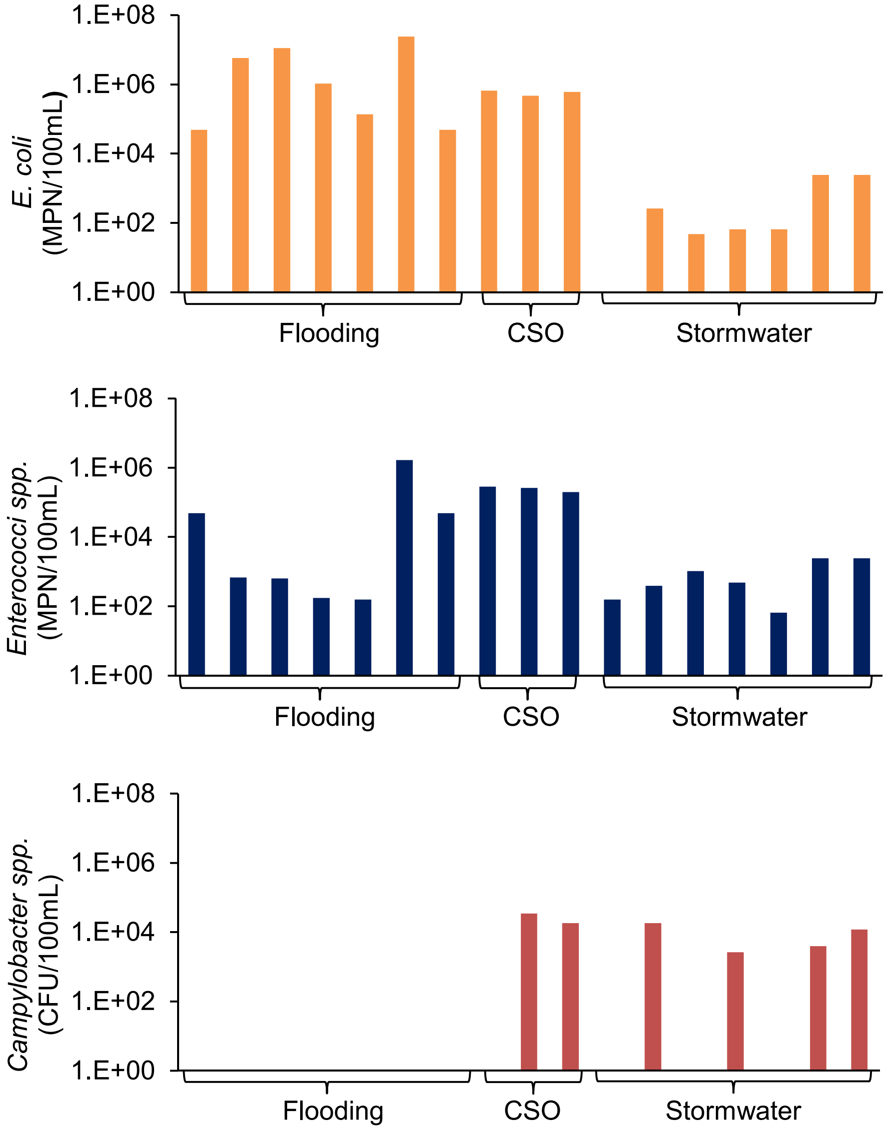

Figure 7. *E. coli*, *Enterococci spp.* and *Campylobacter spp.* concentrations in urban flooding, CSOs and stormwater. Urban floodwater is not analysed for *Campylobacter spp*. For stormwater the missing columns are samples with <1 MPN/100 mL *E. coli* or <1 CFU/100 µl *Campylobacter spp*.

2.3.3 Survival and decay in flooding

Most pathogens originating from humans do not proliferate outside the host, but the time for which they survive, and thereby the degree of decay, depends on flooding conditions. Since flooding happens at any time of the day, flooding scenarios can include outdoor surface flooding with solar radiation (day), cloudy or no solar radiation (night) or indoor flooding without solar radiation.

When investigating the decay of *Campylobacter spp.*, heterotrophs, total coliforms, *E. coli*, *Enterococci spp.* and MS2 Coliphage in floodwater microcosms (wastewater mixed with tap water), for potential outdoor surface flooding, concentrations decreased by 3-5 log units after eight hours of UV irradiation. Decay rate constants were 0.54 to 3.87 h^{-1} at 10 NTU at 0.2 m depth. Furthermore, decay rate constants decreased in line with higher turbidity and depth, probably because high particle content protects microorganisms from solar radiation (Cantwell et al. 2008). In the dark, the decay rate ranged from 0.06 to 0.41 h^{-1} (Andersen et al. II). Decreases in decay were a result of higher turbidity or depth, and minor decay rates in darkness, emphasised the substantial effects of UV radiation on microbial decay.

Outdoor flooding may last several days. In the presence of solar radiation there is potential for solar microbial decay (Figure 8), but still many microorganisms are left in floodwater after eight hours of UV radiation, except for MS2 Coliphage. Eight hours UV radiation was a good representative time frame, because on average, a summer's day in Denmark (May to August) has 7.3 hours of sunlight (DMI data). During the night or indoors, decay is minimal, thereby suggesting that microorganisms can survive for more than one day in flood conditions (Figure 8). In another study made on floodwater in microcosms (wastewater mixed with river water), *E. coli*, *Enterococci spp.*, *Campylobacter spp.* and F-specific phage survived for up to 14 days in semi-darkness (end of the investigation) (Fewtrell et al. 2011). It is therefore likely that microorganisms can survive for longer periods in urban flooding areas, especially in the absence of solar radiation.

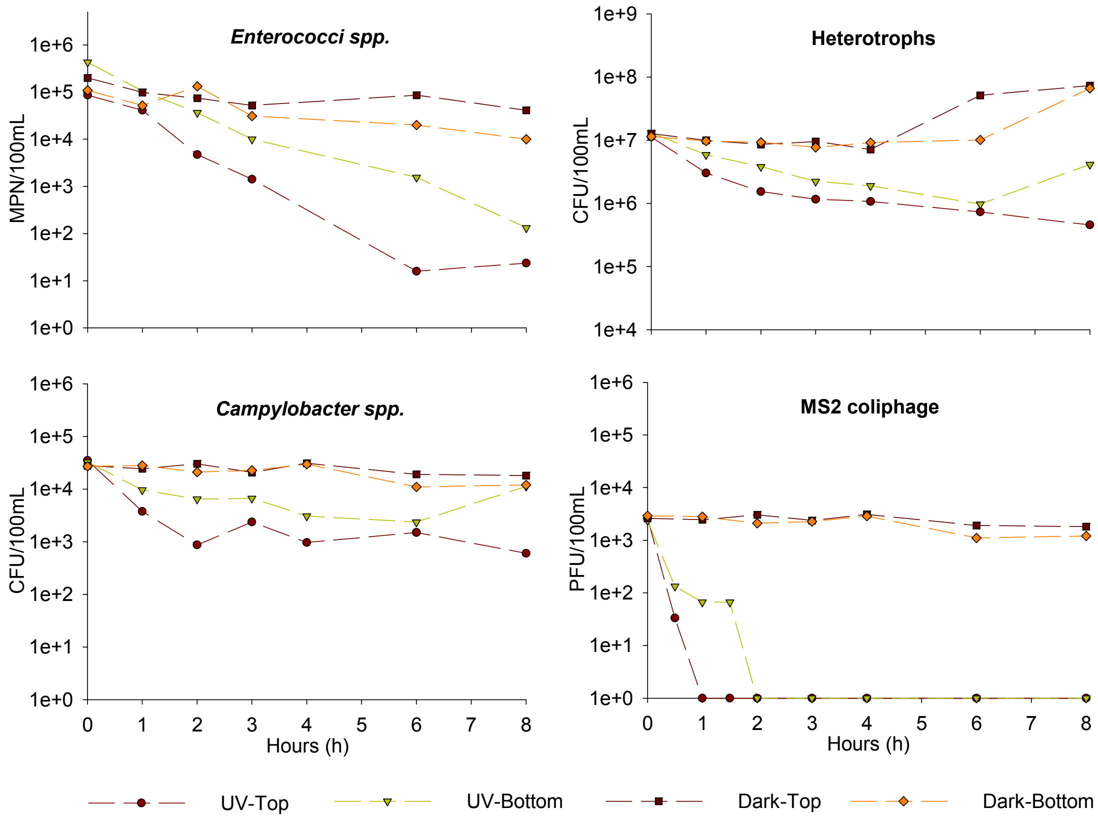

Figure 8. Log-linear curves from floodwater microcosms for decay of the microorganisms: *Enterococci spp*., Heterotrophs, *Campylobacter spp*. and MS2 Coliphage at 10 NTU, 26 W/m^2 or total darkness. Note the Y-axis of the heterotrophs is shown from 10^4 to 10^9 CFU/100mL (Andersen et al. II).

2.4 Data needs for modelling floodwater quality

Modelling microbial concentrations in floodwater is difficult because of the many parameters that cause variations in microbial concentrations, as described in the two previous sections. When aiming at modelling floodwater quality, data requirements are as follows (Figure 1):

- Microbial concentrations in wastewater; input into the model
- Time series of rainfall; input into the model
- Microbial concentrations in CSOs; model validation
- Microbial concentrations in flooding; model validation
- Decay rate constants; modelling changes over time

Measurements are subject to uncertainty, but an average of several measurements (10-24 samples) could be sufficient as inputs (Andersen et al. IV). For model validation, the model results can be compared to CSO measurements, because if they are similar, then the modelling of microbial concentrations in CSOs is accurate, and subsequently it can be assumed that the modelled microbial concentrations in flooding is likewise correct. A final model validation could be based on measurements of microorganisms in flooding.

2.5 Summary of water quality variations

In conclusion, concentrations of pathogenic microorganisms in flooding events vary according to:

- Disease status in the population
- Activity of the community, catchment size and land use
- Dilution in sewers according to rainfall events and the system
- Differences in flooding scenarios and hence different decay rate constants

The measurements from this thesis contributed with valuable data for understanding causes of variations in combined sewers and flooding, leading to a better system understanding.

3 Risk assessment of urban flooding

Forecasting infections caught from exposure to urban flooding, in order to assess human health risks, can not only be achieved by measurements and water quality modelling alone, but also necessitates the use of the quantitative microbial risk assessment (QMRA) framework.

QMRA involves the application of risk assessment principles to estimate the consequences of planned or actual exposure to infectious microorganisms (Haas et al. 1999). The approach includes hazard identification, pathogen fate, transportation in the environment, exposure assessment and microbial dose-response models. Today, QRMA is applied worldwide to establish guidelines and recommendations for the quality of many water types, such as drinking water, bathing water and surface water, to ensure consumer safety and health.

3.1 The concept of QMRA

QMRA quantitatively evaluates risk under defined conditions. The analysis is a calculation by a model, of which the output is the best possible answer to the question of interest. The results of QMRA analysis will therefore always have a substantial level of uncertainty, and there is no single result for the burden of disease.

The QMRA analysis is a dynamic cycle with the key elements (CAC, 1999) (Figure 9):

- *Hazard identification*: a general description of microbial agents and toxins
- *Hazard characterisation*: a description of adverse health effects as a result of exposure to identified hazards in a certain system, to determine the connection between dose and response
- *Exposure assessment*: an estimate of the expected pathogen dose ingested or inhaled, based on the characteristics of the organisms and their survivability. Dose-response models are applied to estimate the probability of infection
- *Risk* characterisation: a approach used to integrate information from hazard identification, hazard characterisation and exposure assessment to estimate the probability of infection or illness in the population. Risk characterisation can lead to risk management decisions.

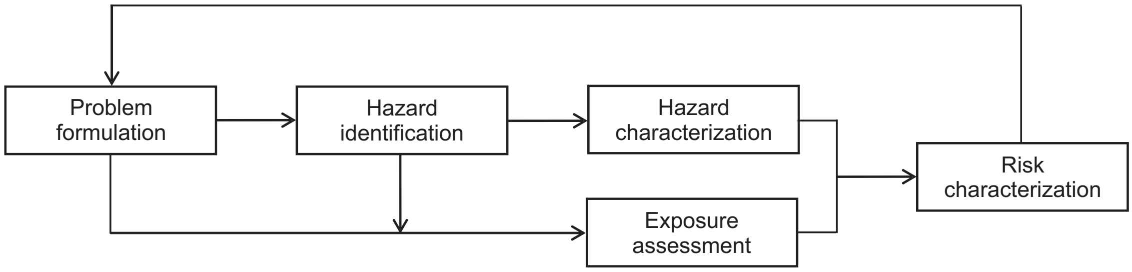

Figure 9. The general steps in a microbial risk assessment, from problem formulation to risk characterization (Modified from Renwick et al. 2003).

3.2 Hazard identification and characterisation

Many highly infectious pathogens can be present in floodwater simultaneously, including bacteria, enteric viruses and protozoa, and it is therefore impossible to quantify the complete spectrum of infections. In urban floodwater pathogens are introduced as a result of wastewater contamination with faecal matter. The large number of pathogens and limited data on most of these means, that it is necessary to be selective about which ones to quantify during the risk assessment process (Fewtrell & Smith 2007). A common approach is to apply a number of reference pathogens usually consisting of a bacterial, viral and protozoan pathogen (WHO 2004). The selection of pathogens can be made by applying the precautionary principle, which presents a worst-case combination of high occurrence, relatively high concentration, high pathogenicity and environmental survival. Also, for QMRA purposes, the pathogens should have established dose-response relationships (Fewtrell et al. 2011, Fewtrell & Smith 2007). The ideal reference pathogens for sewer impacted flooding could therefore be:

- Pathogens associated with transmission through contact with water
- Pathogens with established dose-response relationships
- Pathogens with low infection doses
- Pathogens most likely to cause disease or severe sequelae
- Pathogens occurring in the population
- Pathogens primarily of human origin (urban) and to a lesser extent animals
- The most persistent pathogens
- The pathogen and its occurrence should be well described in the literature

Criteria are applied to find relevant reference pathogens for use in the risk assessments in Andersen et al. (III and IV).

3.3 Exposure assessment

Today, behaviour during flooding and clean-up has been barely documented, even though exposure is unavoidable in nearly all cases. When considering the routes of exposure, the possibilities are ingestion, dermal/wound exposure, inhalation, secondary transmission and mitigating exposure. For urban flooding the most frequent routes of exposure are expected to be through ingestion (faecal-oral route) and skin exposure (Fewtrell & Smith 2007), of which ingestion is typically through hand-to-mouth transfer, though accidental splashing or head immersion may also occur.

Dose-response models are mathematical functions reporting the likelihood of infection by specified pathogen doses (McBride et al. 2013). They are specific for each pathogen and describe the transmission routes and hosts required to define the response, e.g. infection or illness, at a known pathogen dose of a given population (MST 2008, Teunis et al. 1999). For an infection to occur, at least one of the ingested pathogens must survive to start colonisation. This is the basis for dose-response models used for QMRA (Teunis et al. 1999).

Dose-response is described by two models, namely the exponential model (Figure 10) and the Beta-Poisson model (Figure 11). The exponential model is useful when more cells, virions or cysts are needed before infection, while the Beta-Poisson model generally describes the infection when only a few virions or cysts are needed before infection and in many persons, while others are immune.

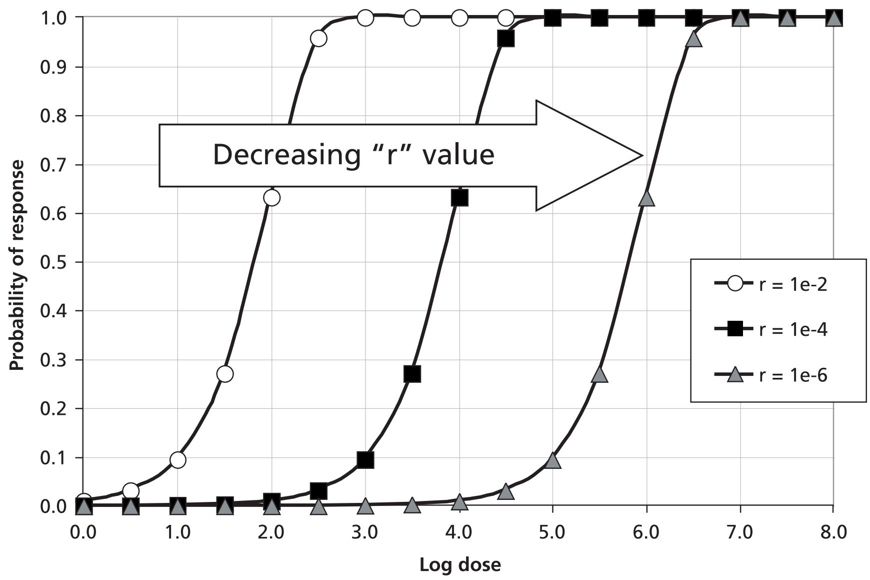

Figure 10. Exponential dose-response functions ($P_{inf} = 1 – e^{-d/r}$) for three different parameter values. The r is a parameter of the dose-response function, which is interpreted as the probability of one cell successfully initiating a response (infection) (FAO, 2005).

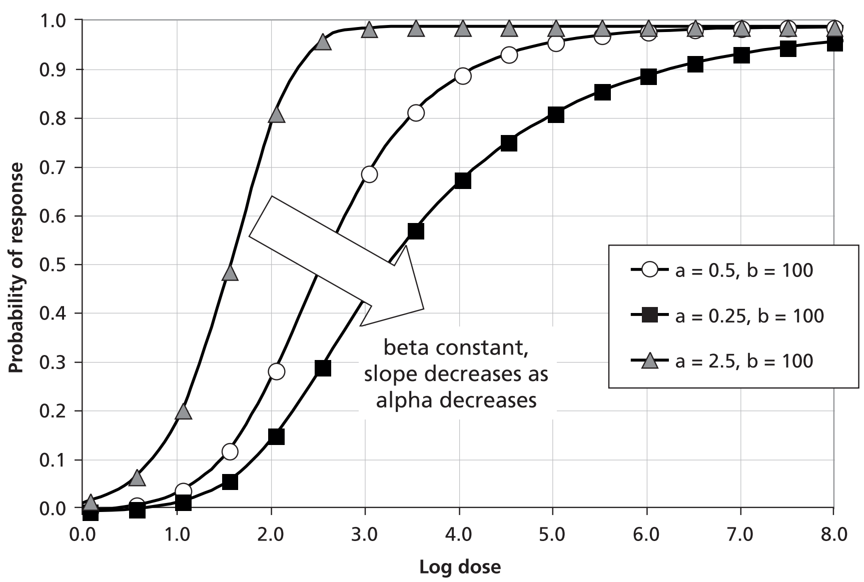

Figure 11. Beta-Poisson dose-response functions ($P_{inf} = 1 – (1 + d/\beta)^{-\alpha}$) for three different parameter values. β and α are parameters of the beta distribution that describes host pathogen interactions (FAO 2005).

3.4 Risk characterisation

The risk characterisation can be used for estimating the burden of disease. When the hazard is identified information is collected through hazard characterisation and exposure assessment, which are integrated into a risk model, exemplified in Figure 1. The risk model is associated with many assumptions and it is therefore very important to state precisely the conditions chosen to estimate the risk, in order to understand uncertainties and make the risk characterisations as reliable as possible, for example if it is a conservative (worst case) or a central estimate (average, median, most likely values) (Arnbjerg-Nielsen 2004). Also the conditions should be stated for making risk assessments comparable between studies of similar scenarios.

3.5 Summary of risk assessment of urban flooding

The quantitative microbial risk assessment framework is a tool used world-wide to predict risk from exposure to contaminated water. Following on from hazard identification the relevant reference pathogens are identified and characterised. The exposure assessment is typically for ingestion risk, because dose-response models are available for this exposure type, though skin contact is also relevant. Finally, a QMRA includes many assumptions, and so the conditions for risk characterisation should be stated very precisely.

4 Limitations of QMRA

The background for estimating risk is the most critical stage in a QMRA. The limitations of the QMRA framework should therefore be identified and managed in order to make reliable risk assessments in the future. There are many approaches available for estimating risk infection from exposure to urban flooding, and there are many limitations of these studies, which ultimately leads to uncertainty in these studies (Table 3).

Table 3. Main articles on microbial risk assessment relating to urban flooding or combined sewer overflows caused by heavy rainfall and sewer overflow. The intention is not to include all existing studies, but to provide knowledge on the key assumptions and limitations of the studies.

Pathogen	Nation	Pathogen concentrations	Dilution in sewers	Dilution on surfaces/ in recipient	Exposure	Ingestion values and type	Dose-response models	Validation Epidemiological studies or measurements	Reference
Fecal streptococcus	US	Measurements of CSO`s	Not included	Not included	Visitor	36mL/d (USEPA, 1997)		Not included	Donovan et al. 2008
Enterococcus					Recreator	7mL/h (USEPA, 2000)			
Giardia					Homeless person	72mL/h (Battelle, 2007)	Exp.[1]		
Campylobacter	NL	Measurements of floodwater	Assumed by comparison of floodwater and wastewater concentrations	Not included	Pedestrian	10 mL for pedestrian (Donovand et al. 2008)	B-p.[2]	Not included	Veldhuis et al. 2010
Giardia							Exp[3]		
Cryptosporidium					Child playing in water	30 mL for children (Schets et al. 2008)	Exp.[4]		
Campylobacter	US	Literature review	Not included	Not included	Recreator	Log-normal distribution for swimmers (Dufour et al., 2006)	B-p.[2]	Comparison to epidemiological studies of swimmers	Soller et al. 2010
Salmonella					Swimmer		Exp.[5]		
E. coli O157:H7							B-p.[6]		
Giardia							Exp.[7]		
Cryptosporidium							Exp.[8]		
Norovirus							B-p.[9]		
Campylobacter	UK	Measurements of floodwater and literature review	Not included	Assumed without scientific explanation	Withdrawal	30mL for adults, 20 mL for children, by head immersion-single gulp swallowed (Westrell et al., 2004)’	B-p.[2]	Not included	Fewtrell et al. 2011
Cryptosporidium							Exp[3]		
Rotavirus					Clean-up	1mL/h (assumption)	B-p.[10]		
Campylobacter		Measurements in streams and literature review	Not included	Log-uniform distribution range beween 1 and 0.01	Swimmer	Ln-normal distribution with mean = 2.92 and stdev = 1.43 (Dufour et al. 2006, Soller et al. 2008)	B-p.[11]	Not included	Viau et al. 2011
Salmonella							Exp.[12]		
Norovirus							B-p.[9]		
Adenovirus							Exp.[13]		
Enterovirus							Exp.[13]		
Salmonella	US	Measurements of CSO`s	Not included	Assumed	Swimmer (primary or secondary contact)	10 mL (Min.), 50 mL (mode), 100 mL (max.) for adults, increased by 50% for children and reduced 80% for secondary contact or inhalation (estimated from literature)	B-p.[14]	Not included	McBride et al. 2013
Norovirus							B-p.[9]		
Giardia							Exp.[7]		
Cryptosporidium							Exp.[15]		
Adenovirus							Exp.[7]		

C. jejuni	NL	Measurements of floodwater	Not included	Not included	Clean-up	0.016 mL for adults		B-p.[16]	Not included	Man
Giardia					Playing	1.7 mL children	Exp[3]			et al. 2014
Cryptosporidium						Found by questionnaire of exposure after flooding events	Exp[3]			
Norovirus								B-p.[9]		
Enterovirus							B-p.[17]			
Vibrio cholera O1	BD	Measurements of wastewater at dry and wet weather	Model	Model	Wading to work or school, exposure via hands	37 mL children lower class (Scheets et al. 2011) 1-7mL children upper class (Dorevitch et al. 2011) Log normal adults lower class (Man et al. 2014) 0.016 mL adults upper class (Man et al. 2014)		B-p.[18]	Not included	Mark et al. submitted
C. jejuni	DK	Literature review	Model	Model	Swimmer	0.60 mL/min for men, 0.44 mL/min for women (Schets et al. 2011)		B-p.[2]	Comparison to an Epidemiological study of the case	Andersen et al. III
E. coli O157:H7								B-p.[6]		
Giardia							Exp[3]			
Cryptosporidium								B-p.[19]		
Norovirus								B-p.[9]		
Campylobacter	DK	Measurements of wastewater, CSO`s and floodwater	Model	Not included	Clean-up	0,016-10mL for adults (Man et al. 2014, USEPA, 2000)		B-p.[2]	Comparison to an Epidemiological study of workers cleaning up. Comparison to meas-urements on floodwa-ter	Andersen et al. IV
Norovirus					Recreator			B-p.[9]		

B-p.: Beta-poisson

Exp.: Exponential

1) Rose et al. (1991), 2) Medema et al. (1996), 3) Teunis et al. (1996), 4) Teunis et al. (1997), 5) U.S. EPA (2006), 6) Teunis et al. (2008b), 7) Haas et al. (1999), 8) U.S. EPA (2006), 9) Teunis et al. (2008a), 10) Haas et al. (1993), 11) Schoen et al. 2010 (modified from Medema et al. 1996), 12) Soller et al. (2008), 13) Haas et al. 1995, 14) Rose & Gerba, (1991), 15) U.S. EPA (2005), 16) Teunis et al. (2005), 17) Teunis and Havelaar (2000), 18) Black et al. (1987), Levine et al. (1981 and 1988), 19) Teunis et al. (2002).

4.1 Pathogen concentrations in flooding

A typical approach taken to identify pathogen concentrations in flooding are measurements from flooding or CSOs, to quantify contamination at the time of sampling (Table 3). When sampling floodwater for microbial analysis, the measurement only gives an indication of floodwater quality at the specific time and place of sampling, and so it is not representative of a larger flooding area. Furthermore, microbial concentrations are determined by the situation, and hence other monitoring strategies, such as monitoring wastewater, may be more useful for predicting microbial concentrations.

4.1.1 Measuring pathogens in floodwater

Measurements of pathogen concentrations in floodwater are useful in risk assessments. These concentrations can be up to 10^6 CFU/L *Campylobacter spp.* and 10^4 Norovirus in floodwater, thus demonstrating that pathogens are present in urban sewers in substantial amounts (Table 2). However, variations in these findings are more than three log folds, demonstrating that the concentration of pathogenic agents is highly variable. In order to understand the causes of differences in measured microbial concentrations, and the reasons for these anomalies, sampling for microbial parameters in floodwater is unsufficient, and so better thought out sampling strategies are required.

When investigating urban flooding in Denmark the sampling approach was sampling floodwater and monitoring oxygen, pH temperature and turbidity on site. Monitoring was useful when evaluating the chances of microbial survival and decay. Information on the duration and origin of the flooding before sampling was collected from residents, in order to identify the reasons for the presence of pathogens. Furthermore, non-microbial parameters such as ammonia and ortho-phosphate were measured, to gain knowledge on wastewater diluted by less polluted rainfall runoff (Andersen et al. IV). The monitoring strategy was thereby improved when compared to sampling and monitoring in other urban flooding risk assessments concentrating only on microbial analysis (Fewtrell et al. 2011, Veldhuis et al. 2010) or are missing measurements of the general floodwater quality (Man et al. 2014).

Measuring physical and chemical parameters can be used to identify causes of variations of microbial concentrations and limitations of analytical methods. For example when testing for Norovirus, the real-time quantitative polymerase chain reaction (PCR) was suspected to have been affected by the high phosphate content (Andersen et al. IV). Unfortunately, physical and chemical parameters are seldomly measured in connection with floodwater sampling.

Collecting information from residents on site is important for identification of the origin and duration of flooding and to understand the causes behind measured pathogen concentrations. One such example could be flooding in a private house, where the residents reported that the washing machine was running at the time of the sewer flooding, which would explain very high phosphate levels (Andersen et al. IV).

Risk assessors typically measure microbial parameters but do not collect data on general floodwater quality and information from residents, and miss details on the origin and duration of the flooding, microorganism decay, inhibition causes and analytical methods. Sampling floodwater is therefore not very useful for predicting floodwater quality in relation to other sources of water, for example rainwater. However, this type of sampling is very important when validating the results of hydraulic water quality models.

4.1.2 Measurements of pathogens in wastewater

In the absence of floodwater samples, wastewater measurements can be used as a reference point for estimation of pathogen concentrations in urban sewer floods (Table 3). But pathogen concentrations in wastewater vary on diurnal basis, depend on the catchment and are inhomogeneous (Andersen et al. I). These variations should therefore be considered when using measurements from wastewater to estimate microbial levels.

The difficulties in integrating diurnal patterns and inhomogeneous distribution can be overcome by use of average microbial concentrations under dry weather conditions. For use in a QMRA the different sampling strategies and measurements for a combined sewer during dry weather conditions could be:

- Diurnal hourly sampling, resulting in 24 samples (Andersen et al. I)
- Diurnal sampling every four hours, resulting in seven samples (Mark et al. submitted)
- Diurnal hourly sampling, but with increased sampling frequency between 6 am and 12 am. (Kim et al. 2009)
- Half-day sampling, 7 am to 6 pm, with a time step of 30 minutes, resulting in 23 samples (Veldhuis et al. 2010)

The high standard deviations found in the studies suggest that when estimating microbial wastewater quality, sampling should be representative and it is therefore most important to collect enough samples, because this could greatly bias interpretation (Lucas et al. 2014). Many of the studies included diurnal sampling, because this provided information on variations throughout the day, including knowledge of when minimum and maximum microbial concentrations occur. Diurnal hourly measurements are also very useful when estimating microbial averages for use as input parameters to model CSOs (Andersen et al. IV).

The conclusion is that the number of wastewater samples for use as background concentrations, to estimate microbial concentrations in flooding, should be representative. For modelling purposes, 24-hour, hourly, diurnal sampling is fitting.

4.1.3 The applicability of pathogen concentrations found in the literature

Literature values for pathogen concentrations in wastewater are useful for estimation of pathogen concentrations in urban sewer flooding in the absence of measurements of floodwater or wastewater. Typically the maximum literature pathogen values in wastewater are used for conservative estimates. When investigating applied concentrations of pathogens from literature studies, the resulting maximum concentrations of different pathogens are similar between studies, thus creating consistency. The applied concentrations are 10^5-10^6 CFU/L *Campylobacter*, 10^5-10^7 CFU/L *E. coli*, 10^2- 10^3 oocysts/L Cryptosporidium, 10^4 cysts/L Giardia and 10^6 PDU/L Norovirus (Andersen et al. III, Soller et al. 2010, Fewtrell et al. 2011).

When aiming at more realistic estimates of the risk of flooding, average or median concentrations of pathogens in wastewater, rather than maximum concentrations, are needed. However, pathogen levels in sewers vary between catchments and nations; hence, it is very difficult to know without measuring which concentrations are applicable for the specific catchment of interest. Literature concentrations are therefore poor for non-conservative estimates relating to the health risks involved in urban flooding.

4.2 Dilution of wastewater as a result of rainfall runoff

To risk assess different levels of rainfall, the ratio of wastewater and rainfall runoff should be addressed. This can be estimated through hydrodynamic models, which unfortunately seldomly are available, and so in most risk assessments the dilution factor is simply assumed or not included (Table 3).

Knowing the dilution factor is not as such the goal of the risk assessment but it is crucial for estimation of microbial concentrations. Time series of rainfall are very useful to estimate the dilution of wastewater by less contaminated rainfall runoff (Andersen et al. III). Time series of rainfall are also useable as inputs into drainage models to estimate the dilutions – and thereby the concentrations – of pathogens (Figure 1) (Andersen et al. IV), as demonstrated for a rainfall event leading to flooding (Figure 12). Thereby is non-scientific assumptions of dilution factors in risk assessments of urban flooding avoided. However, the hydrodynamic models need reliable times series of rainfall inputs, in order to make correct outputs.

In conclusion, estimating the dilution of wastewater according to rainfall runoff for different rainfall events is a barrier in many QMRAs, but this can be overcome by using hydrodynamic models to estimate dilutions, and it is thereby possible to account for spatial and time-dependent variables.

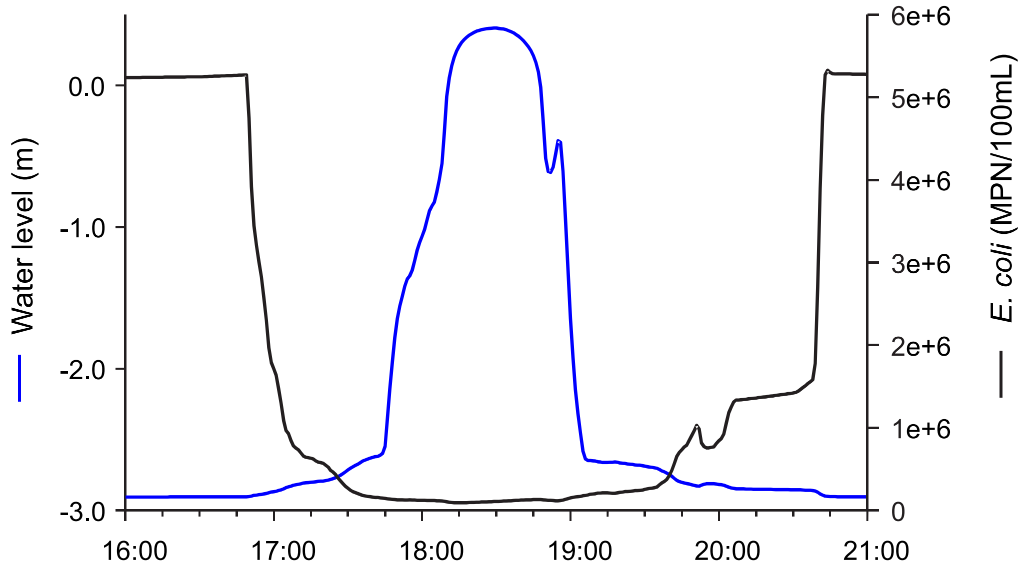

Figure 12. Changes of *E. coli* concentrations in sewers and water levels in a single location during an exemplified model simulation of rainfall. The *E. coli* concentrations decreased when wastewater was diluted by stormwater.

4.3 Exposure assessment

In exposure assessment the exposure scenarios and exposure routes should be defined. Common exposure scenarios associated with urban flooding are swimming, cleaning up, wading, playing and other recreational uses (Figure 13).

Figure 13. Examples of different exposure scenarios: swimming, cleaning up and wading through flooding. (Photo: S. T. Andersen)

The exposure route considered in an urban flooding QMRA is typically ingestion. Knowing ingestion volumes when estimating infection risk from exposure to flooding from combined sewers is very important, because this will highly influence the results of the risk assessment (Figure 14) (Man et al. 2014).

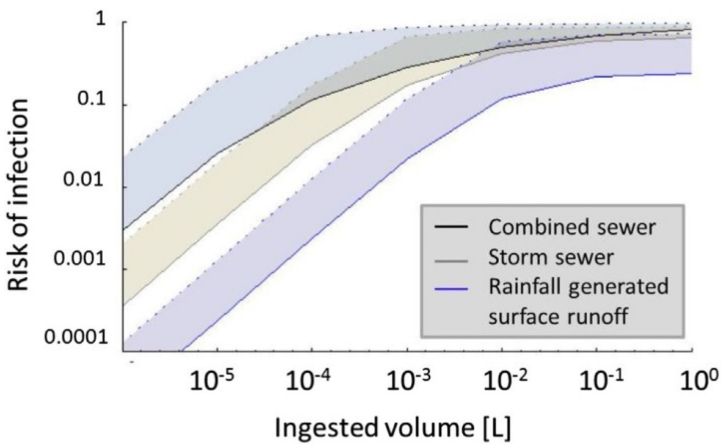

Figure 14. Mean risk of infection for exposure to floodwater originating from combined sewers, storm sewers and rainfall-generated surface runoff, as a function of the ingested volume per exposure event (95th percentiles are shown by dotted lines) (Man et al. 2014).

In a study of cleaning up basements, an ingestion volume of one mL/hour was used, without any scientific explanations (Fewtrell et al. 2011). For recreational scenarios, the ingestion volumes were estimated to be near seven mL/h (Donovan et al. 2008) and 10 mL per event (Veldhuis et al. 2010), which were estimated by USEPA risk assessment guidance for individuals exposed to surface water when wading (USEPA, 2000). A recent study investigated water intake in relation to flooding through a questionnaire. The mean volume was 1.7 mL (confidence interval 0-4.6 mL) for children exposed for 21 minutes and 0.016 mL (95% confidence interval 0-0.068 mL) for adults exposed for 18 minutes (Man et al. 2014), which are very low ingestion volumes compared to the previous estimated intakes in the mentioned risk assessments, suggesting that some earlier risk assessments overestimate the risk of infection.

Knowledge of ingestion volumes together with exposure duration allows for cumulative dose estimations, thereby estimating the doses received over the entire exposure period. Unfortunately, even though the study by Man et al. (2014) considered the purpose of exposure, the results were not reported in the paper, which is a major limitation when other risk assessors need to apply data from the study. Also, ingestion volumes found from the findings of questionnaires are uncertain, because people assume intake levels but it is difficult for them to truly know this figure. Nevertheless, the study by Man et al. (2014) contributed valuable data on ingestion volumes from exposure to urban floodwater, although more profound investigations of ingestion volumes are still required.

In conclusion further data of ingestion volumes from exposure to urban floodwater are needed. Also integrating exposure data is very subjective in QMRA studies of urban flooding, and there is no common approach for making exposure scenarios or using literature values when estimating ingestion volumes.

4.4 Dose-response for ingestion of floodwater

In microbial risk assessments of urban flooding, Beta-Poisson models are typically applied for *Campylobacter* and Norovirus, whereas exponential models are generally applied for Giardia and Cryptosporidium (Table 3). For *Campylobacter*, studies apply the same dose-response model for adults and for children (Table 3). For Norovirus, studies apply the only existing dose-response model which is based on the surrogate Norwalk virus. For Giardia and Cryptosporidium, different exponential dose-response models are applied, often without reporting the basis for deciding on these particular models (Table 3). In one study, however, the use of dose-response models for Giardia and Cryptosporidium were restricted to those that could be derived mathematically from fundamental principles defined in the literature, rather than empirical decisions (McBride et al. 2013).

The available dose-response models are mostly for healthy adults (Parkin et al. 2003), which are not very representative of children and other susceptible parts of the population. Nonetheless, some studies use dose-response models for estimating the risk of infection for children in the absence of specific models (Mark et al. submitted, Fewtrell et al. 2011). This may result in an underestimation of the infection; even so, they are adjusted for differences in ingestion volumes and exposure times.

Dose-response models apply to specific species or geno-groups of the pathogens and often relate to high pathogen doses (Besner et al. 2011). However, surrogate organisms are not always similar to the measured organisms. For example, dose-response models for *C. jejuni* are applied for *Campylobacter spp*. (Andersen et al. IV, Fewtrell et al. 2011) and dose-response models for Norovirus GI are applied for Norovirus GII (Andersen et al. IV, Man et al. 2014, Viau et al. 2011), in the absence of correctly fitting dose-response models for the species or genogroup in question. This type of application causes uncertainty in the use of dose-response models and may result in lower or higher risks than actuality, depending on use.

Thus the use of dose-response models is consistent between QMRA studies, because only a few dose-response models exist for each reference pathogen. The dose-response models are, however, limited in use because they are mostly for adults and for specific species or geno-groups of pathogens, which are not always the ones analysed in risk assessments. Nevertheless, establishing new dose-response models is not an easy task, because it is ethically challenging and because it requires very controlled experiments.

4.5 Validation of risk assessment

A microbial risk assessment is always subjected to a substantial uncertainty, and hence a validation step would improve the values of the assessment. The validation could be a comparison of estimated risk of infection to the risk of illness estimated in epidemiological studies or by comparison of estimated pathogen concentrations to measurements taken from a flooding event.

A risk assessment of swimming in polluted bathing water was validated with the results of an epidemiological study of swimmers from the same case (Andersen et al. III). From another risk assessment, investigating the health effects from exposure to urban flooding in the capital of Denmark, the validation was a comparison to the results of an epidemiological study of professionals cleaning up in the same case, but also included a comparison to the estimated risks found by flooding measurements (Andersen et al. IV). The result was that the estimated infection was close to the reported levels of illness, thereby demonstrating the value of including hydrodynamic water quality models in estimation of risk of infection from exposure to CSO, and thereby potential flooding.

Except for these two studies (Andersen et al. III and IV), it is not typical to evaluate by directly comparing to epidemiological studies of the case, or the validation step is left out because the risk assessment is a fictitious case, or because the assessment is for real cases but epidemiological data are absent (Table 3).

Since the detection of pathogens is not directly associated with infectivity, exposure to pathogens does not always result in infection, and neither does infection necessarily progress to clinical illness (WHO 2005, Teunis et al. 1999). The risk of infection is therefore higher than the illness rate, and hence any comparison to epidemiological studies is only an indication of the relevance of the risk assessment, and thus not a total solution. To overcome this issue the risk of infection can be adjusted to the risk of illness (Fewtrell et al. 2011), but such an adjustment is yet another uncertainty factor added to the risk assessment.

To sum up, today most of the risk assessments of urban flooding are not validated because of the absence of epidemiological studies, as they are either fictitious cases or floodwater measurements are basic and therefore not very useful for validation.

4.6 Comparison of estimated risks to guidelines for drinking water

Applying risk assessment in decision making would be useful, but so far this is not implemented in relation to urban flooding, recommendations or guidelines on “acceptable” urban flooding risks. The criterion for drinking water, for which the most highly exposed populations should not exceed a risk level of 10^{-4}, is a benchmark set up by the US Environmental Protection Agency

(USEPA). This criterion could be a substitute for criteria in urban flooding because guidelines or recommendations are useful when aiming at using QMRA results in flood management solutions. The criterion for drinking water, set out by the USEPA, may be too strict in relation to urban flooding because the risk of infection is typically $>10^{-4}$ from exposure to urban flooding for otherwise healthy adults and children, and thus urban flooding would be a substantial problem. Only the risk of infection by *Vibrio cholerae* in a case study of flooding is below the guideline level (Table 4). On the other hand, the criterion is useful because it demonstrates that urban flooding is a significant hazard to public health, and solutions should be implemented to avoid this type of flooding.

Table 4. Examples of estimated single exposure infection risks for urban flooding and CSO situations.

Events	**Impacted waters /locations**	**Estimated risks**	**Pathogens**
Combined sewer overflows	Bathing water	10^{-1}	*Campylobacter*, *E. coli O157:H7*, Norovirus, Giardia, Cryptosporidium[a)]
	River water	10^{-2}	Giardia[b)]
	Streams/Cannals	10^{-2}-10^{-4}	Giardia and Cryptosporidium[c)]
Urban sewer flooding	Properties and surfaces	10^{-3} to 10^{-1}	*Campylobacter spp.*[d)]
	Properties and surfaces	10^{-5} to 10^{-1}	Norovirus[e)]
	Properties and surfaces	10^{-5} to 10^{-1}	*Enterococci spp.*[f)]
	Properties and surfaces	10^{-2} to 10^{-1}	*Giardia* and Cryptosporidium[g)]
	Surfaces	10^{-6} to 10^{-5}	*Vibrio cholerae*[h)]

a) Andersen et al. III, b) Donovan et al. 2008, c) Shets et al. 2008, d) Andersen et al. IV, Veldhuis et al. 2010, Man et al. 2014, e) Andersen et al. IV, Man et al. 2014, f) Andersen et al. IV, g) Man et al. 2014, h) Mark et al. Submitted.

4.7 Summary of QMRA limitations

The many assumptions in urban flooding QMRAs increase the uncertainty of any resulting risks. From investigations of the different approaches of QMRA or urban flooding, the limitations are identified as:

- Low numbers of pathogen concentration measurements in floodwater and/or wastewater
- Knowledge gap in estimating dilution in sewers and on the surface
- Knowledge gap regarding ingestion volumes, according to exposure scenarios
- Limited usability of available dose-response relationships
- Low numbers of epidemiological studies of urban flooding events
- Absence of guidelines

In conclusion, QMRAs can estimate the risk of infection from exposure to urban flooding, but there are still knowledge gaps and hence it is possible to improve the QMRA framework to make even better risk assessments of urban flooding.

5 Improving QMRAs

Current risk assessments have many limitations, in that risk assessors have to make many assumptions. Fortunately, there are options for improving many of these limitations (Table 5).

In this PhD thesis the approach is to improve an approach to better predict pathogen concentrations in urban flooding for use in microbial risk assessments. This is done by improving the number of quantitative data for use in hydrodynamic water quality models. Improving knowledge of dose-response models and ingestion volumes, and validation by epidemiological studies specific to urban flooding, is also needed but is beyond the scope of this thesis.

Table 5. Limitations and possible improvements to urban flooding QMRAs.

Parameter	Limitations	Possible improvement
Data collection and pathogen concentrations	Lack of knowledge of variations of the microbial concentrations in wastewater	Sampling of wastewater that includes considerations of the diurnal and catchment variations
	Sampling of floodwater is only representative for the time and location of sampling, the situation and rainfall	Use hydrodynamic drainage and flood models to make estimations for larger areas and different rainfalls
	Low knowledge of origin and duration of flooding's, leading to low knowledge of causes of microbial decay	Collect information from residents on site of flooding and make measurements of general floodwater quality
Dilution in sewers and on surfaces	Absence in use of hydrodynamic water quality models	Use of hydrodynamic drainage and flood models to estimate dilutions
Exposure	Low numbers of data of ingestion volumes, that are related to defined exposure scenarios for urban flooding	Make questionnaires to identify the possible ingestion volume, exposure time, reason for exposure and type of exposed persons
Dose-response models	Mostly for healthy adults, less often for children and elderly	Studies of dose-response for subpopulations such as children and elderly to know differences in dose-response as compared to adults
	For specific species or genogroups of pathogens	Studies of the results from applying dose-response models for other species or genogroups of pathogens than the ones specifically belonging to the dose-response studies
	Absence of dose-response models for skin exposure	Studies of dose-response for skin exposure
Validation	Low numbers of epidemiological studies related to urban flooding in combination to risk assessment	Epidemiological studies of flooding incidents that includes most of the questions from the proposed questionnaire for exposure
	Low numbers of measurements of pathogens in flooding	Sampling of floodwater

5.1 Improving risk assessments by using measurements and hydrodynamic water quality models

It is a novel approach to use hydrodynamic water quality models to estimate the risk of infection from urban flooding. Previous risk assessments of urban flooding used measurements from flooding, but their results were influenced by circumstances, for single rainfall events and locations. Hydrodynamic water quality models are therefore very valuable in urban flooding QMRAs, because they are useful for many different rainfall levels and locations during flooding. Also, uncertainties associated with dilution in sewers, and potentially on the surface, are solved by using hydrodynamic models. However, using these models requires reliable quantitative data on rainfall volumes and microbial concentrations, and measurements are needed for validation.

5.1.1 Proof of concept – risk assessment of CSO contaminated bathing water

The approach in this thesis considers that if the risk of infection from swimming in CSO polluted bathing water can be estimated by using hydrodynamic water quality models, then it should also be possible to estimate the risk of infection for estimated dilutions of wastewater as a result of urban flooding. We therefore assessed the risk of CSO-contaminated bathing water, using two dynamic models. A drainage model was used to estimate dilution in sewers, and a 3D hydrodynamic bathing water model was used to estimate the dilution of CSO water in bathing water (Figure 15). Both models were water quality models that can potentially estimate pathogen concentrations, but in the absence of quantitative data the models were only used for estimating dilutions in sewers and bathing water. The input parameters used to estimate dilution from sewers and into bathing water were times series of rainfall and CSO discharge data. Data on literature-based concentrations of five reference pathogens were applied in the absence of measurements. The resulting risk was estimated for an ironman competition that took place in CSO-polluted bathing water (Andersen et al. III). There was a close similarity between estimated infection risk and reported illness, both being 42%, which demonstrated that the approach of applying hydrodynamic models to estimate health risks from overloaded sewers, in this case a CSO incident, with input data from time series of rainfall and pathogen concentrations, was useable.

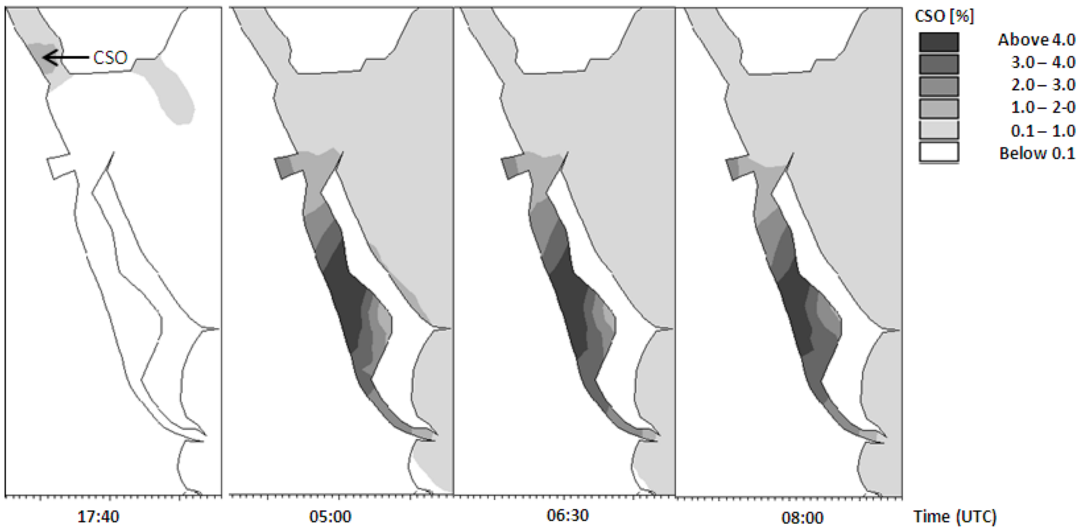

Figure 15. Model output of the percentage CSO water in bathing water, August 14-15th 2010, start of overflow (17:40), and start (5:00), middle (06:30) and end (08:00) of the ironman competition. UTC; Universal Time Coordinated (Andersen et al. III).

5.1.2 Application of approach – risk assessment of urban flooding

In a risk assessment of urban flooding, measurements were used in combination with hydrodynamic water quality models. The pathogen concentrations in urban flooding and thereby the risk of infection from exposure were estimated by a drainage model. Time series of rainfall and an average of pathogen measurements from a diurnal sampling of wastewater were used as inputs into the drainage model, in order to compute dilution in sewers. The drainage model was validated by comparison of the model results to measurements in CSOs with a very good agreement (Figure 16) (Andersen et al. IV). Because microbial concentrations in CSOs can be modelled, even if there is very high dilution in sewers during rainfall, the model simulates dilution correctly (Figure 16). The assumption is therefore that if we can model microbial concentrations in CSOs then we can also model microbial concentrations in flooding, even though there is more dilution during flooding than for CSOs.

The risk of infection was 10^{-3} to 10^{-1} from exposure to flooding, estimated for the two reference pathogens *Campylobacter* and Norovirus and by using thoroughly considered literature data relating to ingestion volumes from exposure to flooding, including cleaning up and wading (Andersen et al. IV). The estimated risks were validated by comparing the estimated risk from measurements of flooding with epidemiological studies, which were similar, and this demonstrated the applicability of the approach.

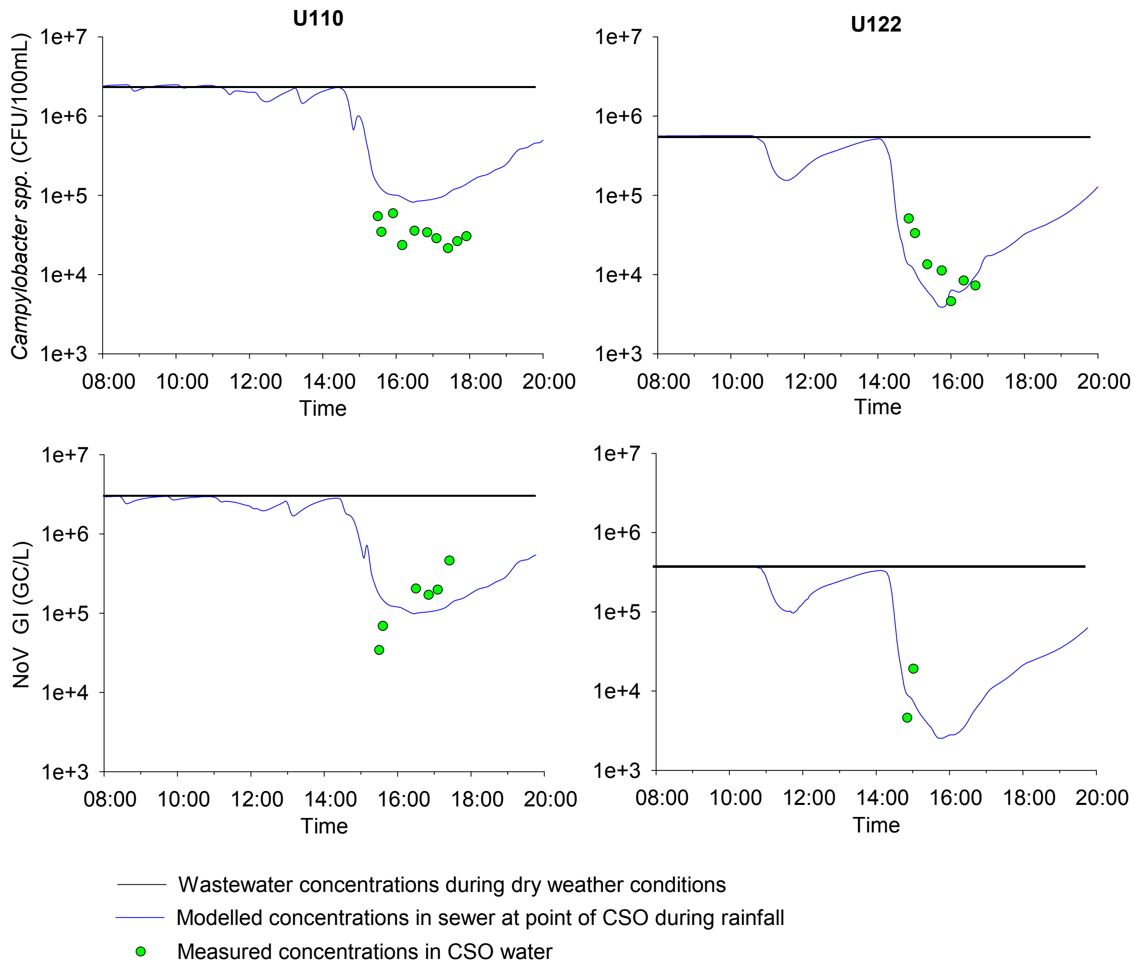

Figure 16. *Campylobacter* and Norovirus GI in wastewater during dry weather conditions and from the CSO event at CSO structures U110 and U122. Note: For Norovirus only quantified data are shown (Andersen et al. IV).

5.1.3 Improving the approach – integrating decay rate constants

When estimating microbial concentrations in surface flooding, microbial decay should be considered, because if microbial concentrations decrease over time, then the risk of infection is lowered. In this thesis microbial decay was integrated manually after modelling floodwater quality to estimate pathogen concentrations in flooding over time (Figure 1). Ideally, though, decay should be included in a surface model, and currently such surface models are being developed. One such application was a risk assessment of urban flooding in Bangladesh where, a 1D-2D coupled surface model was used to model flood extent, transport and dilution of sewage water on the surface (Mark et al. submitted). However, the model did not consider microorganism decay, mainly because data were missing on the decay of pathogens in this setting.

This thesis, however, contributed valuable data on microbial decay in urban flooding situations, which could be integrated into hydrodynamic models. More data are still needed, though, in order to determine the full extent of decay and the reasons for the decay over time of pathogens in urban flooding.

5.2 Identification of improvements and recommendations

This PhD thesis has identified the limitations of and possible improvements to urban flooding QMRAs through a literature review of the current approaches to risk assessment and through the work presented in the four papers included herein. Some of the suggested improvements are integrated into urban flooding risk assessment in Andersen et al. (IV).

5.2.1 Improvements resulting from the thesis

The approach of using pathogen concentration measurements in hydrodynamic water quality models, in order to estimate the risk of infection from exposure to urban flooding, has many advantages compared to previous risk methodologies (Table 6). The major improvements deriving from this thesis are:

- Measurements of microorganisms in wastewater, CSOs and flooding, along with measurements of physical and chemical parameters, provide essential data, not only to identify the concentrations of microorganisms in flooding, but also to identify the causes of these concentrations. As hydrodynamic water quality models rely heavily on quantitative data, such measurements are very important.
- Microbial decay rate constants can be determined for urban flooding situations. In flooding situations, it is clear that decay should be included in urban flooding QMRAs, because microorganisms change significantly in numbers over time.
- Using hydrodynamic water quality models in association with measurements is very useful for urban flooding QMRAs. Unfortunately the use of such models is limited, mainly because of lack of availability. When using hydrodynamic water quality models, the risk of infection is not only estimated for different rainfall levels but also for spatial points during flooding, and the dilution is known, thereby avoiding being subject to change over time and assumptions regarding dilution. It is therefore possible to apply an intelligent reduction of urban flooding to reduce public health

risk, for example by focusing on locations with high concentrations and exposure, when sewers are being developed.

Table 6. Improvements to urban flooding QMRAs as a result of this thesis.

From measurements	From drainage model
Knowledge of concentrations of microorganisms in wastewater, CSO`s and flooding	Possibility of estimating risks for different rainfalls
Data and knowledge of variations of microbial concentrations in wastewater, both diurnal and in catchments	Knowledge of dilution by rainfall runoff
Knowledge of general floodwater quality and causes of measuring high or low microbial concentrations in flooding	Knowledge of spatial distribution of flooding
Knowledge of decay of microorganisms in flooding	Knowledge of duration of flooding

5.2.2 Recommendations

The work of this thesis has led to the following recommendations, when aiming at using measurements for modelling floodwater quality for use in urban flooding QMRAs:

- Average values of measured microbial concentrations in wastewater during dry weather flow are the best choice for a central estimate, because of diurnal variations of microbial concentrations in sewers. The maximum measured microbial concentrations should be supplemental, in order to achieve a conservative estimate of the risk of infection from exposure to urban flooding.
- Catchment variations should be considered because there are considerable differences in microbial concentrations between catchments. It should be determined why catchments have similar or different microbial concentrations, but until then, data from one catchment are not transferable for another catchment.
- Information on the duration and origin of flooding should be collected from residents when collecting floodwater samples, in order to gain knowledge on the origin of flooding and the reasons for measured pathogen levels. Also, as a supplement to measuring microbial parameters, water quality parameters, such as turbidity, oxygen, temperature, conductivity

and pH, as well as UV radiation, can be measured. These measurements provide additional information on the conditions relevant for survival and decay and can be easily measured at the site of flooding. As another supplement, chemical or physical parameters, such as suspended solids and ammonia, could be measured because they also indicate the origin of flooding.

6 Conclusions

This thesis has presented an overview of the limitations of and possibilities of improving urban flooding QMRAs. The overall strategy was to lower the degree of assumptions on microbial concentrations by using measurements as a validation step. The main conclusions were:

Limitations of urban flooding QMRAs

Limiting parameters for making urban flooding QMRAs were identified as:

- Too few pathogen measurements, both from flooding and wastewater
- Uncertainty relating to ingestion volumes, because of the very few studies on this topic
- Limited usability of dose-response models, because they are for specific genotypes and species and mostly for adults and not more sensitive groups
- Limited knowledge of the dilution factor, both in sewers and on surfaces (ratio of wastewater and rainfall runoff)
- A low degree of validation because of the lack of measurements of microbial concentrations in flooding and epidemiological studies.

Improving the knowledge of ingestion volumes requires studies specific to exposure scenarios for urban flooding. Improving knowledge on the usage of dose-response models in relation to urban flooding requires studies that have some ethical considerations. Improving knowledge on ingestion volumes and dose-response models is therefore very difficult, but measuring microbial concentrations in wastewater and flooding was possible, and estimating the dilution factor was completed in this thesis.

Measurements of microorganisms in sewers and floodwater

The data basis was improved through measurements of microbial concentrations in wastewater, combined sewer overflows and floodwater. The work demonstrated that urban floodwater quality varies according to:

- The prevalence of pathogens
- Variations in wastewater quality
- The rainfall event and following dilution in sewers
- The survival and decay rate constants of microorganisms

For variations in wastewater quality, catchment variation was the most important, whereas the diurnal pattern was less important. During flooding the measured levels of indicator bacteria and pathogens were substantial, and pathogens were often found in connection with urban flooding. The microbial concentrations, however, changed over time because of the microbial decay caused by UV radiation, whereas decay was minor in darkness. This knowledge could be incorporated into a QMRA when estimating risk of infection.

Modelling microbial concentrations in flooding

It was possible to estimate microbial concentrations in flooding by using hydrodynamic water quality models in combination with measurements. This approach was innovative when compared to previous approaches, which mostly considered risk in relation to flooding measurements.

The drainage model was very useful for simulating the dilution and transport of microorganisms in sewers, which were found by comparing modelled and measured microbial concentrations in CSOs and floods. Hydrodynamic models are therefore very useful for predicting the risk of infections related to future rainfall. The reliability of the model, though, still relies heavily on the input data.

Improving urban flooding QMRAs

From a combination of measurements and the use of a drainage model it was possible to make an urban flooding QMRA. Measurements of microbial concentrations in wastewater were used as inputs into the drainage model, in order to estimate microbial concentrations during flooding. The model results were validated by comparing them to measurements from CSOs and floodwater and to reported illnesses in epidemiological studies.

The model was run for a heavy rainfall event in Denmark. The results showed a substantial risk of infection, being 10^{-3} to 10^{-1}, from exposure to urban flooding, which was comparable to the risk estimated from measurements of microbial concentrations.

This thesis demonstrated that the approach of combining measurements with hydrodynamic water quality models was useable for estimating microbial concentrations, and thereby useful in a QMRA. Furthermore, this approach could supplement decision support for different rainfall scenarios and locations in urban areas with combined sewers.

7 Perspectives

The knowledge gained in this thesis could be used for future microbial risk assessments of flooding in urban areas. This thesis shows the importance of assessing the limitations leading to uncertainty in this respect, which is important to know when aiming at making realistic QMRAs.

From this thesis, several challenges were identified when determining microbial concentrations in urban flooding for use in risk assessments, and these resulted in the following suggestions for future research:

- To know exactly what determine differences between catchments, more measurements and detailed knowledge of the catchments are needed. As a result, it could be possible to generalise microbial concentrations in specific catchments, which in turn will be useable for modelling purposes. Since microbial analyses can be very expensive, sensors for parameters such as ammonia, which fluctuate less than microbial parameters but originating from faeces, just like microorganisms, could be used in pre-investigations into general wastewater quality. Relevant catchments for measuring microbial parameters could thereby be identified.
- In the study on survival and decay, the effects of UV radiation, depth and turbidity were investigated. Other parameters that could be relevant to investigate are the effects of nutritional conditions and whether predation contributes to decay observed in dark conditions.
- Methods used to quantify microorganisms in microbial risk assessments of flooding generally do not distinguish between infectious and non-infectious pathogens. As it is possible that some of the pathogens lose infectivity outside the host, this infectivity should be investigated to avoid overestimating risk.
- In this work a drainage model was validated through measurements of CSOs. However, the model was not validated through comparison to measurements of flooding in catchments, which is essential in this developing phase to make sure that the model's results are realistic.
- Improvements of exposure assessment by more studies of ingestion volumes from exposure to urban flooding. The studies should clearly describe the type, time and cause of exposure, along with intake volumes, in order to make generalise intake findings.

- In this work a drainage model was applied, while survival and decay were determined for different scenarios. The next step would be to combine the output of the drainage model with a surface model (flood model), to model surface water, for which decay data would be highly relevant. Surface models are currently being developed and some are ready for use; hence, they will soon be used in urban flooding QMRAs.

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9 Papers

I **Andersen, S. T.**, Mark O., Sharma, A. K., Schultz, A. C., Albrechtsen, H.-J. Diurnal variations of *E. coli*, *Enterococci spp*. and Norovirus in wastewater at different catchments at dry weather flow: measured and modelled. *Submitted*.

II **Andersen, S. T.**, Mark O., Albrechtsen, H.-J. Survival of microorganisms in urban floodwater from overloaded combined sewer systems – effects of UV irradiation simulating sunlight. *Submitted*.

III **Andersen, S. T.**, Erichsen, A. C., Mark O., Albrechtsen, H.-J. (2013) Effects of a 20 year rain event: a quantitative microbial risk assessment of a case of contaminated bathing water in Copenhagen, Denmark. *Journal of Water and Health*, Vol. 11 (4), p. 636-646.

IV **Andersen, S. T.**, Mark O., Schultz, A. C., Albrechtsen, H.-J. Improvements of quantitative microbial risk assessment of urban flooding by combining quantitative microbial data with a drainage model *Manuscript*.

In this online version of the thesis, the papers are not included but can be obtained from electronic article databases e.g. via www.orbit.dtu.dk or on request from.

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